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Session II. Economic Methodologies and Ecological Constraints: Case Studies of Marine Pollution

Figure 4

Damages from Oil Spills

as a Function of Quantity Spilled

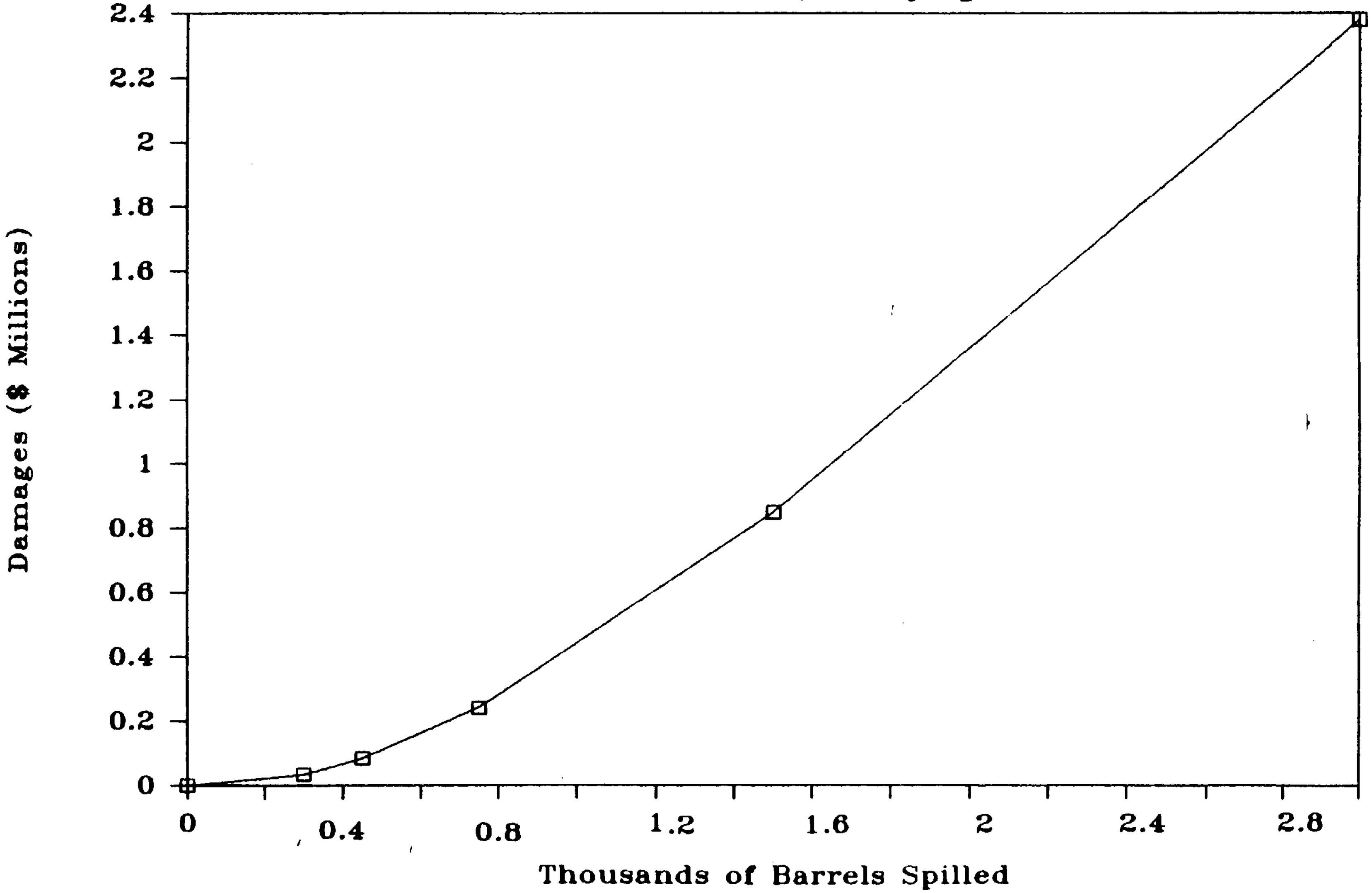

Table 5 Total Damages to All Categories from a 75 Metric Ton Spill of Prudoe Bay Crude Oil (20% Volitiles) in an Estuarine, Intertidal Environment in the Virginian Province

	Sandy Shoreline	Rocky Shoreline
Fishery Losses	$ 42.	$ 80035.
Bird and Fur Seal Losses	$ 192.	$ 1754.
Damages to Public Beaches	$ 40481.	$ 0.
Total for All Categories	$ 40715.	$ 81789.

Empirical stuff is incorrect

REGULATION OF MARINE CONTAMINATION UNDER ENVIRONMENTAL UNCERTAINTY: SHELLFISH CONTAMINATION IN CALIFORNIA

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REGULATION OF MARINE CONTAMINATION UNDER ENVIRONMENTAL UNCERTAINTY: SHELLFISH CONTAMINATION IN CALIFORNIA[1]

Contamination of marine resources by industrial, agricultural and municipal wastes is a key environmental concern. Pollution of marine systems impairs both the quantity and quality of services provided. Commercial and sport fish and shellfish harvests may be reduced because of gradual increases in pollution levels or chemical spills. Consumption of fish and shellfish may become more risky because of bioaccumulation of heavy metals and pesticides or because of microbial contamination from sewage and agriculture. Populations of other species may decline, sometimes to the point of extinction. Recreational waters may become unswimmable and recreational areas unusable because of water contamination or oil/chemical spills.

Regulation pertaining to these problems aims at achieving some sort of balance between the value of the activities creating the pollution and that of the marine resources affected. Thus, an assessment of the tradeoffs between polluters' production, the quantity of marine resource output (fish harvests, number of recreation sites, species diversity), and the quality of marine resource output (human health risks from fish or water contamination) is essential for policy determination. One of the central problems in obtaining accurate assessments of these tradeoffs is that there is typically a great deal of uncertainty about the environmental effects. The ways in which contaminants enter and disseminate through the environment tend to be monitored and modeled only incompletely. Stochastic factors such as weather patterns also tend to have a large degree of influence on them. Many of the fundamental mechanisms underlying physiological responses - both in humans and animals - are not understood well at this time. Moreover, many physiological responses have a multiplicity of causes and contributing factors, making attribution of causality a severe problem.

The general attitude toward these contamination problems emphasizes prevention: policy is increasingly expected to be prospective rather than reactive. This orientation forces regulators to rely on simulation models about whose structure and parameters there is usually considerable uncertainty. Also, the public tends to be quite sensitive to severe outcomes - for example, cancers, birth defects, large wildlife kills or outbreaks of food poisoning - even when these outcomes are rather unlikely. Thus, regulatory decisions need to be based on methodologies which incorporate uncertainty explicitly.

In an earlier paper (Lichtenberg and Zilberman 1985) we proposed a method for constructing uncertainty-adjusted cost curves for environmental risk reduction as a means of meeting this need. This paper applies that approach to a case of human health risk from microbial shellfish contamination. We begin with a discussion of our approach, it uses and its characteristics. We then take up the case of contamination of oyster beds in Tomales Bay, California by dairy runoff. We consider the tradeoff between closing the fishery and requiring the construction of holding ponds to prevent runoff of dairy wastes as mechanisms for reducing the risk of gastroenteric illness from oyster consumption. We then examine the differences in total cost between standards based on mean risk versus uncertainty-adjusted risk and between cost-efficient requirements for holding pond construction versus requirements which are uniform across dairies. Finally, we consider the implicit value of risk reduction under different regulatory approaches.

Constructing Uncertainty-Adjusted Cost Curves for Environmental Risk Reduction

Our approach to constructing uncertainty-adjusted cost curves for risk reduction essentially involves combining a probabilistic environmental risk assessment - a type of risk assessment that has become increasingly widespread in recent years - with a safety rule decision mechanism. The resulting estimates of uncertainty-compensated tradeoffs between risk and social cost can then be used for policy determination using formal decision criteria (cost-benefit, risk-benefit) or for subjective evaluation of regulatory alternatives.

This procedure has a number of appealing characteristics. First, it makes use of the full range of information available in a practical manner. Even though all of the uncertainty present cannot generally be parameterized, it is important to incorporate as much as possible into regulatory decisions. Second, this approach is more amenable to interdisciplinary cooperation. It utilizes the kinds of information produced by risk analysis and is equivalent to using confidence intervals for statistical decision making, the method preferred in the natural sciences. Third, because safety rules have been used in a variety of economic applications, they are well understood. Moreover, they have been shown to approximate well expected utility decisions in a number of empirical contexts (Thomson and Hazel1 1972). Fourth, the safety rule approach corresponds quite closely to the terms of much of the relevant legislation, which requires regulators to provide adequate protection to public health or the environment within a sufficient margin of safety. It also corresponds to a "disaster-avoidance" approach to decision making which is widespread among the public and the regulatory community. Thus, our approach can be said to suit the preference structure of decision makers.

A set of uncertainty-adjusted cost curves for risk reduction can be derived as follows. Assume that N regulatory instruments $X_1,...,X_N$ are available and that the extent to which each is applied can be measured as an increasing function to social cost. Social cost will include such items as costs of decontaminating the environment, costs of installing exposure reduction devices, welfare losses sustained by consumers and producers due to regulation-induced productivity decreases/cost increases, and government expenditures on monitoring and enforcement. Environmental risk is assumed to be a function of policy, R(X), in a manner defined by a environmental risk assessment. Let R_0 denote a the maximum allowable risk (risk standard) and P represent the frequency with which this standard is met (margin of safety). The regulatory problem can be defined as choosing a policy mix $X_1^*,...,X_N^*$ to minimize the cost of ensuring that the risk standard R_0 is maintained with a given margin of safety P. This problem can be written formally as

(1) $\min X_1+...+X_N$

subject to the constraint

(2) $Pr[R(X) \leq R_0] \leq 1-P$.

Repeated application of this procedure letting R_0 vary parametrically and keeping P fixed yields a set of minimum cost tradeoffs between the risk exceeded only with frequency 1-P, denoted R(P), and the total social cost of regulation $X^*=x^*_1+...+X_N^*$ this set amounts to an uncertainty-adjusted cost curve for risk reduction $X^*[R(P)]$. Repeating this procedure over the range of P yields a complete set of such cost curves.

Note that the margin of safety, P, essentially represents the decision maker's aversion to uncertainty: the larger P is, the greater will be the emphasis on the right hand tail of the risk distribution and hence the greater will be the weight placed on uncertainty in choosing a mix of regulatory instruments.

In our earlier paper, we analyzed the optimal mix of regulatory instruments for the risk model proposed for cancer risk assessment by Crouch and Wilson (1981) and subsequently adopted for that purpose by the U.S. Environmental Protection Agency. This model assumes that risk can be expressed well by a multiplicative combination of parameters and that the risk exceeded with any given probability can be represented by a weighted sum of the mean and standard deviation of the risk.[2] We showed that the optimal policy will consist of a portfolio of measures, some specializing in reducing mean risk and others specializing in reducing uncertainty about risk. Emphasis on reducing uncertainty becomes greater as aversion to uncertainty grows, possibly even to the point where performance with respect to average risk actually declines. Emphasis on mean risk reductions is greater for more toxic contaminants, less controllable risks and for cases where the background uncertainty is large.

The set of optimal policy portfolios for all levels of risk and a given margin of safety P makes up a cost curve for risk reduction with margin of safety P. It is decreasing in risk, i.e., downward sloping in cost-risk space. An increase in toxicity, in uncontrollability or in background uncertainty will increase cost, i.e., push this curve up and to the right. Cost curves associated with higher margins of safety (greater aversion to uncertainty) will lie up and to the right as well.

The slopes of these cost curves, given by the set of lagrange multipliers associated with the constraint (2), represent the marginal cost of reducing risk with a given margin of safety. As such, they give at least a lower bound estimate of social willingness to pay for risk reductions with any given level of aversion to uncertainty. Thus, they can be used to construct a measure of the implicit value of risk reduction with a given margin of safety. In our analysis of the Crouch and Wilson risk model, we showed that this marginal cost of risk reduction increases as aversion to uncertainty increases and as risk, toxicity, uncontrollability and background uncertainty decrease.

These cost curves can be used to generate regulatory decisions using formal methodologies or subjective comparisons of alternatives. For example, cost-benefit solutions can be derived for any given margin of safety by equating marginal benefit and marginal cost; risk benefit solutions can be derived by requiring that the risk/cost ratio be consistent with historical experience (Starr).

Microbial Contamination of Shellfish in Tomales Bay

Pollution has had a severe impact on shellfisheries throughout the U.S. Shellfish harvests have declined significantly because of pollution, overfishing and natural phenomena. Many fisheries have been closed permanently for public health reasons; many other are subjected to recurring temporary closures for the same cause (Larkin and Hunt 1982).

In earlier times, the main public health concern was the spread of typhoid from consumption of sewage-contaminated shellfish. More recently, the focus of concern has shifted to hepatitis A and other viral diseases, although salmonellid gastroenteritis is still a significant problem. Alarm over the health risks from consumption of raw shellfish has heightened in recent months. For example, a recent editorial in the New England Journal of Medicine urged that people cease eating raw shellfish altogether because the risks of gastroenteric illness are simply unacceptable (Du Pont 1986). This sentiment is shared by much of the medical community (Eckholm 1986).

Tomales Bay, located about 50 miles north of San Francisco, houses a small cultivated oyster fishery. The main threat to the fishery comes from effluent runoff from the surrounding dairy industry. Under normal conditions, runoff from the manure spread over each dairy's disposal area will have a negligible effect on water quality and hence on the fishery. During severe rainstorms, though, the manure will be washed into the watershed. The resulting fecal contamination of the bay will lead to contamination of the oyster beds. If the water quality around the oyster beds fails to meet public health standards, the fishery will be closed. Current standards, expressed in terms of the most probable number (MPN) of bacteria, are median values of 70 MPN/100 LM water for total coliform bacteria and 14 MPN/100 LM water for fecal coliform bacteria (Hunt 1977).

Contamination can be reduced by constructing holding ponds at the dairies. Each pond will retain effluent up to a fixed capacity. Because the manure floats, it will be washed into the watershed as soon as that capacity is exceeded. The severity and frequency of pollution will thus depend on the capacities of the holding ponds built. Current regulations require all dairies to maintain facilities to hold a 10-year 24-hour rainfall in addition a maximum average rainfall of 5.75 inches occurring in the previous three weeks (Rafter et al. 1974).

Hochman, Zilberman and Just (1977) analyzed optimal regulation of the Tomales Basin under the assumptions of zero acceptable risk and of uniformity of holding ponds capacities across dairies. In this case pollution is all-or-nothing: either the bay is contaminated and the fishery shut down or the bay remains clean and the fishery stays open. Their analysis suggested that, to maximize the sum of fishery and dairy profits, dairies should be required to construct ponds capable of holding 50-year 24-hour rainfall events so that the fishery would be closed only one-fifth as often as under current regulations.

This characterization of the situation has several shortcomings. First, it ignores heterogeneity among dairies. Because of topoographical differences, the costs of constructing a holding pond of any given capacity tend to vary widely; Hochman, Zilberman and Just's data indicate that they may differ by a factor of four or more. This suggests that any given standard can be achieved at a lower overall cost by discriminating among dairies, i.e., requiring dairies in more favorable locations to build large holding ponds and those in less favorable locations to build smaller ones. Second, it examines only a simple, very stringent risk standard - essentially one of zero acceptable risk. This implies that the fishery must be closed whenever pollution is present. Thus, the analysis ignores the possibilities of substitution between closing the fishery and building holding ponds, which may permit standards to be achieved at lower cost. Moreover, it would be useful to examine the choice of a standard as well. For this purpose, it would be preferable to generate costs for a range of standards. Third, because the risk is all-or-nothing, uncertainty enters in a very restricted way, namely as the probability that a risk occurs. Examination of a broader range of standards would have to incorporate much more of the probability distribution characterizing the risk. Fourth, only one source of uncertainty - rainfall - is modeled. There is also significant uncertainty about other factors affecting risk, including the amounts of dairy waste present, the limits between runoff and microbial contamination of the bay, the links between water quality and shellfish contamination and dose-response relations for illness resulting from the consumption of contaminated shellfish.

This study attempts to overcome some of these shortcomings. In what follows, we combine Hochman, Zilberman and Just's data on runoff control costs with a model of acute gastroenteritis risk from raw oyster consumption. We employ the approach described in the previous section to derive sets of uncertainty-adjusted, cost-efficient tradeoffs between total control cost and risk which we then use to analyze alternative regulatory approaches.

Risk of Acute Gastroenteritis from Microbial Shellfish Contamination

For simplicity, the risk of acute gastroenteritis from microbial shellfish contamination was modeled as a multiplicative combination of variables reflecting water quality, microbial uptake by oysters, and human physiological response. Two policy instruments were included: (1) a requirement that each dairy build a holding pond of a designated size, not necessarily identical, and (2) closure of the fishery.

Each cow was assumed to produce an identical constant amount of manure and each pound of manure was assumed to have a constant, identical microbial content. The microbial content of manure was assumed to be measured, accurately by the coliform count. Manure in runoff was assumed to be mixed uniformly in the bay waters around the oyster beds. This is undoubtedly an oversimplification, but does constitute a reasonable first approximation (see for example the data presented by the Northeast Technical Services Unit 1980).

Under these assumptions, water quality around the oyster beds will be proportional to the number of cows generating effluent. Runoff is affected by two things: (1) the sizes of holding ponds at the dairies, which is a function of policy, denoted X_1 and (2) rainfall, denoted R. Write the size distribution of holding ponds (that is, the number of dairies having holding ponds of capacity k) as $W(k, X_1)$ and the set of dairies generating runoff for any given 24-hour rainfall level r as $B[W(k,X_1),R]$. If there are J dairies having herds of sizes $H_1,...,H_J$, water quality can be written as $q\sum_{j\varepsilon B} H_j$, where q is a constant of proportionality.

To estimate q, it was assumed that during extremely heavy rainstorms effluent runs off at all dairies and that this effluent is the sole source of microbial contamination as measured by the fecal coliform count. This ignores the contributions from human sources such as defective septic tanks, which are important in certain areas but of secondary significance overall. The maximum fecal coliform count measured over a broad range of oyster beds was 16,000 MPN/100 ml water (Northeast Technical Services Unit 1980). The total number of cows at dairies in the watershed, as reported by Hochman, Zilberman and Just was 13,200, implying a value of q = 1.2 MPN/100 ml water per cow.

Uptake of microbial contaminants by the oyster population was assumed to be proportional to water quality around the oyster beds. Letting u denote this uptake parameter, the proportion of oysters contaminated can be written as $uq \sum_{j \epsilon B} H_j$. This parameter was estimated using data presented by Andrews et al. (1975) in their study of fecal coliform counts as an indicator of bacteriological contamination. They reported percentages of oyster samples testing positive for salmonella for five ranges of fecal coliform counts. These percentages were regressed on the midpoints of the fecal coliform count ranges without a constant term, with an additional assumption that the maximum observed fecal coliform count was 16,000 MPN/100 ml water, to obtain a estimate of u = 0.000057. The standard deviation of this estimate was 0.000022.

This approach has several shortcomings. First, it assumes that the presence of salmonella in oyster mean indicates a sufficient level of microbial contamination to cause illness. It would be preferable to measure the extent to which the meat is contaminated as well; however, the literature on dose-response relations looks only at whether or not the shellfish are contaminated, so this additional level of detail would add little of substance to the analysis. More significant is the implicit assumption that the presence of salmonella is also a good indicator of viral contamination. The results of a number of studies indicate the contrary, specifically that the fecal coliform count in surrounding waters tends not to be such a good indicator of viral contamination (see for instance Gerba et al. 1980). However, no better indicator seems to be available at present.

A number of studies have shown that bacterial and viral populations remain relatively constant after processing and cold storage for several weeks (Kelley and Arcisz 1954; Wilson and McCleskey 1951; Erickson, Vasconcelos and Presnell 1967; Di Girolamo, Liston and Matches 1970; Hood, Baker and Singleton 1984). Because this study is concerned with incremental risk from water contamination, additional contamination from shucking and handling and improper storage were ignored.

The probability of contracting gastroenteritis from eating contaminated oysters was assumed to be proportional to the proportion of oysters contaminated. Letting d represent the dose-response parameter, this probability can be written as $duq \sum_{j \in B} H_j$. Variations in this dose-response relation due to variations in the number of oysters eaten or the extent of contamination of each oyster were ignored. No data on bacteriological poisonings from consuming contaminated shellfish were known to us, so epidemiological studies of viral infections were used exclusively. Several such studies investigated incidents where guests at large parties had been served contaminated shellfish. The proportion of people consuming these shellfish who contracted gastroenteritis ranged from .36 to .65 (Gill et al. 1983; Gunn et al. 1982; Morse et al. 1986). The dose-response parameter d was given a value .45, in accord with the incidence of gastroenteritis reported by Gunn et al. and the combined incidence of gastroenteritis and hepatitis reported by Morse et al.

Finally, the risk of gastroenteritis may be eradicated by closing the fishery and preventing harvest of the shellfish until they have decontaminated themselves. This effect can be represented by a function $o(X_2,)$ which takes on a value of one when the fishery remains open and zero when the fishery is closed.

The risk of acute gastroenteritis from consumption of contaminated oysters, denoted t, can thus be written as t = $o(X_2)duq\sum_{j\varepsilon B} H_j$, a function of policies relating to the fishery, X_2, and to the construction of holding ponds, X_1.

As a first step, we will consider only one source of uncertainty, namely the randomness of 24-hour rainfall. Other sources of uncertainty will be incorporated in a future study. The probability distribution of 24-hour rainfall was modeled as a Pearson type II distribution as in Hochman, Zilberman and Just. Using data from the California Department of Water Resources (1974) they estimated that the 24-hour rainfall exceeded with probability b was 2.928 + K(b)0.413, where K(b) was chosen to correspond to a Pearson type III distribution with a skrewness of 1.3. Letting F(r) denote the probability that 24-hour rainfall R does not exceed a given level r, the risk associated with any given level of 24-hour rainfall can be written:

$$t = [1-F(r)]o(X_2)duq \sum_{j \varepsilon B[W(k,X_1),r]} H_j.$$

Calculation of Cost Curves for Gastroenteric Risk Reduction

Numerical methods were used to estimate total social costs and probability distributions of risk under alternative policies and to determine the minimum cost policy mix for each level of uncertainty-adjusted risk. Three levels of aversion to uncertainty were analyzed: neutrality, which implies regulation based on mean risk, and safety margins of 95% and 99%.

The Tomales Bay fishery produces a small share of total oyster production in the San Francisco Bay area: loss of the harvest was thus assumed to have a negligible impact on prices. The analysis performed by Hochman, Zilberman and Just suggested that holding pond construction costs would never be great enough to force dairies out of business, so this potential impact was ignored. It was assumed also that periodic c1osure of the fishery would have negligible long run impacts on production or employment. This characterization seems reasonable in light of the fact that most growers are small, individual operators using a minimum of equipment, so that entry is quite inexpensive. Finally, it was assumed that monitoring and enforcement costs are negligible, which also seems reasonable in light of the fact that the California Department of Health Services has only one person responsible for shellfish in all of Northern California. These assumptions imply that total expected social cost is closely approximated by the sum of amotized holding pond construction costs and expected losses in revenue from closure of the fishery.

The probability distribution of 24-hour rainfall events was partitioned into a number of discrete segments. For each of these states of nature the data collected by Hochman, Zilberman and Just were used to estimate the cost at each dairy of building a holding pond with capacity sufficient to hold no more than the corresponding 24-hour rainfall given the average maximum rainfall of 5.75 inches in the preceding three weeks. Following Hochman, Zilberman and Just, total construction costs were annualized by multiplying them by an interest rate of 10%. The result of these calculations was a set of functions for each dairy $C_1(b),..., C_J(b)$ representing the annualized cost of building a holding pond with sufficient capacity for 24-hour rainfalls exceeded only 100b% of the time.

Expected revenue losses from closure of the fishery were estimated by multiplying the frequency of closures under alternative policies times the annual revenue of the fishery. Following Hochman, Zilberman and Just, fishery revenue was estimated at $500,000 per year. No adjustment for inflation was made in either case; thus, all costs are expressed in terms of 1973 dollars.

Costs and risks under a policy approach relying only on the construction of holding ponds under neutrality toward uncertainty were estimated as follows. Each dairy was assumed to behave as if it faced a tax T on effluent (effectively, on herd size). Constructin a holding pond with a capacity exceeded only with probability b will cost $C_j(b)$. The expected tax payments in this case will amount to bTH_j. Minimization of the sum of these two costs implies that the optimal capacity for the j^{th} dairy, b^*_j, should be set to equate the marginal construction cost $C'_j(b^*_j)$ with the maximum tax liability TH_j. Construction cost for the j^{th} dairy is then $C_j(b^*_j)$ and total construction cost is found by summing up construction costs over all dairies. The choice of a capacity also gives a two-point distribution of effluent: effluent is zero with probability $1-b^*_j$ and proportional to H_j with probability b^*_j Aggregation over the full set of dairies gives a probability distribution of total effluent which was then transformed into a probability distribution of risk using the model described in the previous section. Mean risk was calculated from that probability distribution, for each tax level, giving a risk, cost pair. Repeating this procedure over a range of values of T gave a set of such pairs, i.e. a cost curve for risk reduction.

For policies based on 95% or 99% margins of safety, the same procedure was followed with the addition of a constraint that holding pond capacity never be large enough for 24-hour rainfalls exceeded less frequently than the margin equal to P, this meant $b^*_j \leq 1-P$. The measure of risk used under these policies was the level exceeded only 100(1-P)% of the time, 5% and 1% for the 95% and 99% margins of safety, respectively.

The combined impact of the two policies of building ponds and closing the fishery was estimated using the same procedures with the additional constraint that pond capacity never exceed the frequency with which the fishery would be closed. Thus, if the closure policy were to prevent harvest after 24-hour rainfalls no less than 100-yearevents, dairies were constrained to build ponds with capacities no greater than 100-year 24-hour rainfalls. If the fishery were closed with probability c, this implied a constraint of $b^*_j < c$ for neutrality with respect to uncertainty and $b^*_j < 1\text{-}P\text{+}c$ for a margin of safety P. The total cost of these policies was found by adding expected revenue losses to total construction costs as described above.

The efficient mix of policies and minimum-cost tradeoff curves for each level of aversion to uncertainty were estimated by comparing the total cost curves generated by these different closure rules. The resulting social cost curves for risk reductions were the lower envelopes of the total cost curves for each of these rules.

For policies under neutrality with respect to uncertainty, the cost of achieving a mean risk standard using uniform holding pond construction requirements was estimated by finding the frequency with which the maximum risk level would have to occur to generate the same average risk. Letting t* denote this maximum risk level, the capacity required to achieve any level of average risk t is $b^*u = t/t^* + c$, where c is the probability that the fishery is closed. Total construction costs for this uniform capacity bu* and revenue losses from closure were then calculated and compared as described above.

Such fine tuning is not possible for policies under 95% and 99% margins of safety: risk is either at a maximum or zero. The key problem in such cases is calculating the minimum cost combination of closure and construction required to achieve zero risk. This was done by comparing the total cost curves for alternative closure policies using uniform building requirements.

Empirical Results

Some preliminary results of this analysis are shown in Figures 1-4.

Consider first the choice of policy instruments between closing the fishery and requiring construction of holding ponds. As Figure 1 shows, closing the fishery as seldom as 1% of the time is suboptimal except for very low risk standards. This occurs because closing the fishery is a relatively expensive way to reduce risk. It costs $5000 to reduce risk by 1% by closing the fishery. Building holding ponds becomes that costly only at very large capacities. For example, Hochman, Zilberman and Just found that, under a uniform building requirement, the marginal costs of these two instruments were equal only when holding ponds were capable of holding up to 50-year 24-hour rainfalls. The buildining requirements derived here are even less costly, implying that the marginal costs of the two instruments will be equated only at even larger capacities. Expressed in terms of risk, Figure 1 indicates that closing the fishery for 100-year and larger 24-hour rainfalls in cost-efficient under neutrality toward uncertainty only for risk standards of .001 and less.

The optimal policy mix changes as aversion to uncertainty increases. Figures 2 and 3 compare fishery closure with efficient building requirements under margins of safety for 95% and 99%, respectively. Closure is not used under a 95% safety margin. This occurs because 24-hour rainfalls larger than 20-year events are of no concern to policy makers at this level of aversion to uncertainty. As we have seen, fishery closure becomes cost-competitive with building requirements only at much larger capacities. In contrast, closure is under for much higher risk standards under a 99% safety margin than under neutrality toward uncertainty. This occurs because very large, infrequent 24-hour rainfalls count very heavily under this level of aversion to uncertainty.

Now consider the impact of aversion to uncertainty on the cost curve for risk reduction. There are two exceptions to this. First, the 95% and 99% margin of safety cost curves are identical at maximum risk. This occurs because the construction of holding ponds has no effect on risk with these margins of safety until capacities of 20-year and 100-year 24-hour rainfalls are reached. Second, mean risk standards becomes more expensive than 95% margin of safety standard at very low risk levels. The reason for this is that decisions made under a 95% safety rule ignore risks that occur less frequently than 5% of the time, i.e., when very large rainstorms occur. Thus, a "zero" risk level is attained even though some residual risk actually occurs (albeit infrequently). It is precisely these residual risks which are the most expensive to eradicate. Under mean risk oriented regulation, their share of weight in decision making grows as the overall risk level declines, so that regulation becomes increasingly costly. These low-frequency risks do play an important part regulation with a 99% margin of safety, though, with the result that the cost of mean risk standards never exceeds the cost of 99% margin of safety standard.

Excluding these two cases, it can be seen that the premium paid for reductions in uncertainty declines as the risk standard decreases. For example, achieving a 99% margin of safety costs about $10,000 more than achieving a 95% margin of safety for higher risk levels. This implies an average "uncertainty premium" of about $2500 for each 1% increase in the margin of safety. For very low levels of risk, though, this difference decreases. The gap between cost under neutrality toward uncertainty and under a 95% margin of safety is even greater for higher risk levels, about $80,000 for a risk of 0.35, for example. It decreases much more rapidly, though, and declines to about $5000 for very low risk levels. Neutrality toward uncertainty in this model corresponds to a safety margin of 45%, implying that the "uncertainty premium" averages about $1,600 for each 1% increase in the margin of safety for very high risk levels and falls as low as $100 for very low risk levels. This suggests that the cost of aversion to uncertainty increases as the level of aversion to uncertainty rises, a result consistent with the standard literature on risk and uncertainty.

Finally, consider the increase in social cost from the use of uniform holding pond size requirements such as are now in force. Preliminary calculations suggest that the gap between the minimum cost of risk reduction and cost under a uniform building policy is greatest at high risk for all levels of aversion to uncertainty and that the two converge as the risk standard approaches zero. The latter result occurs because, to attain a zero risk standard, it is generally necessary to require holding ponds of the maximum relevant capacity, effect a uniform building requirement. Under neutrality with respect to uncertainty, the additional social cost of a uniform policy is about $55,000 at high risk levels, declines rather rapidly to $2,000 as risk declines and disappears as risk approaches zero. For safety margins of 95% and 99%, this additional cost reaches only about $7,500 and $5,000 respectively, suggesting that efficiency losses decline as aversion to uncertainty increases.

Concluding Remarks

We have argued that uncertainty tends to matter a great deal in environmental policy problems - including those concerning marine pollution - because of the incompleteness of the knowledge base from which decisions must be made and because of the expressed preferences of the general public. This implies in turn that formal methods for making environmental policy decisions should incorporate uncertainty explicitly. We have proposed once such methodology, which applies a safety rule decision mechanism to a probablistic model of environmental risk to generate uncertainty-adjusted cost curves for risk reduction. We argued that this approach combines several theoretically appealing features and practical usefulness.

The empirical analysis of microbial contamination of oysters in Tomales Bay by dairy wastes bears out these contentions. Calculation of the uncertainty-adjusted cost curves was feasible and the curves behaved as expected: they were decreasing in risk and increasing in aversion to uncertainty. Aversion to uncertainty was shown to have a significant influence on the choice of policy instruments: closing the fishery turned out to be cost-efficient only for very high levels of aversion to uncertainty.

The empirical example also demonstrated some interesting characteristics of decisions made in this fashion. The premium in social cost required to achieve reductions in risk and increasing in aversion to uncertainty.

Regarding pollution control measures, the analysis suggested that inefficiencies due to the imposition to uniform control technologies tend to be greatest for high risk levels and low aversion to uncertainty. This result has some interesting implications for broader pollution control policies.

In sum, we believe that the analysis shows that our proposed methodology can be useful tool for environmental decision making and yields some interesting perspectives on a number of aspects of environmental policy questions.

Footnotes

[1]Although the information described in this article has been funded wholly or in part by the United States Environmental Protection Agency under assistance agreement CR811200-03 to the Western Consortium for the Health Professions, it has not been subjected to the Agency's required peer and administrative review and therefore does not necessarily reflect the views of the Agency and no official endorsement should be inferred. We would like to thank Dave Alton, Doug Price, Bob Hultquist and Mel Yee for their assistance. Responsibility for any errors is, of course, ours.

[2]This assumption is not very restrictive. Such representations can be derived for any probability distribution using Chebyshev's inequality or the partial moment methods proposed by Berck and Hihm (198?) and by Atwood.

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Figure 1
Construction vs. Closure: Mean Risk

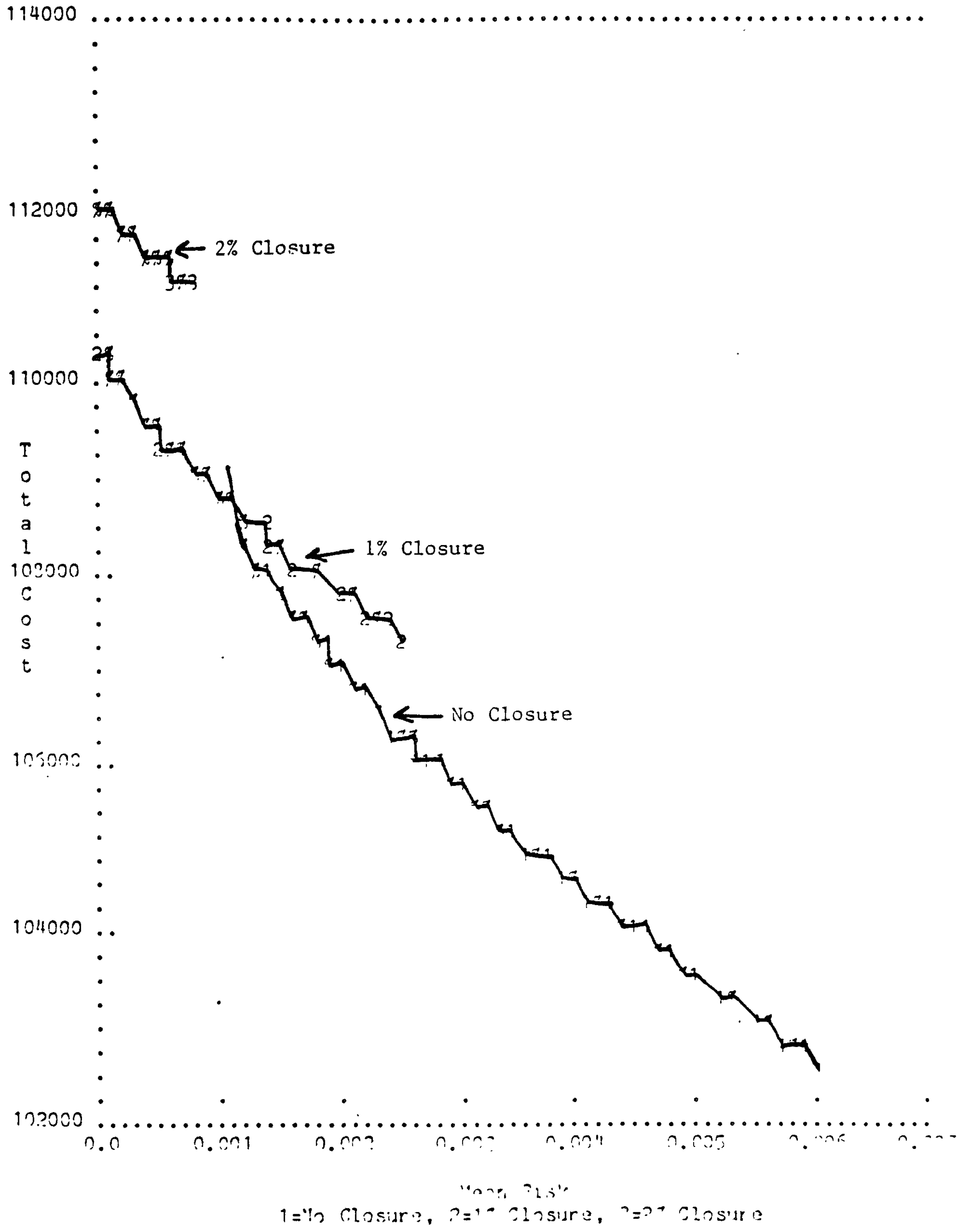


Figure 2

Construction vs. Closure: 05% Safety Margin

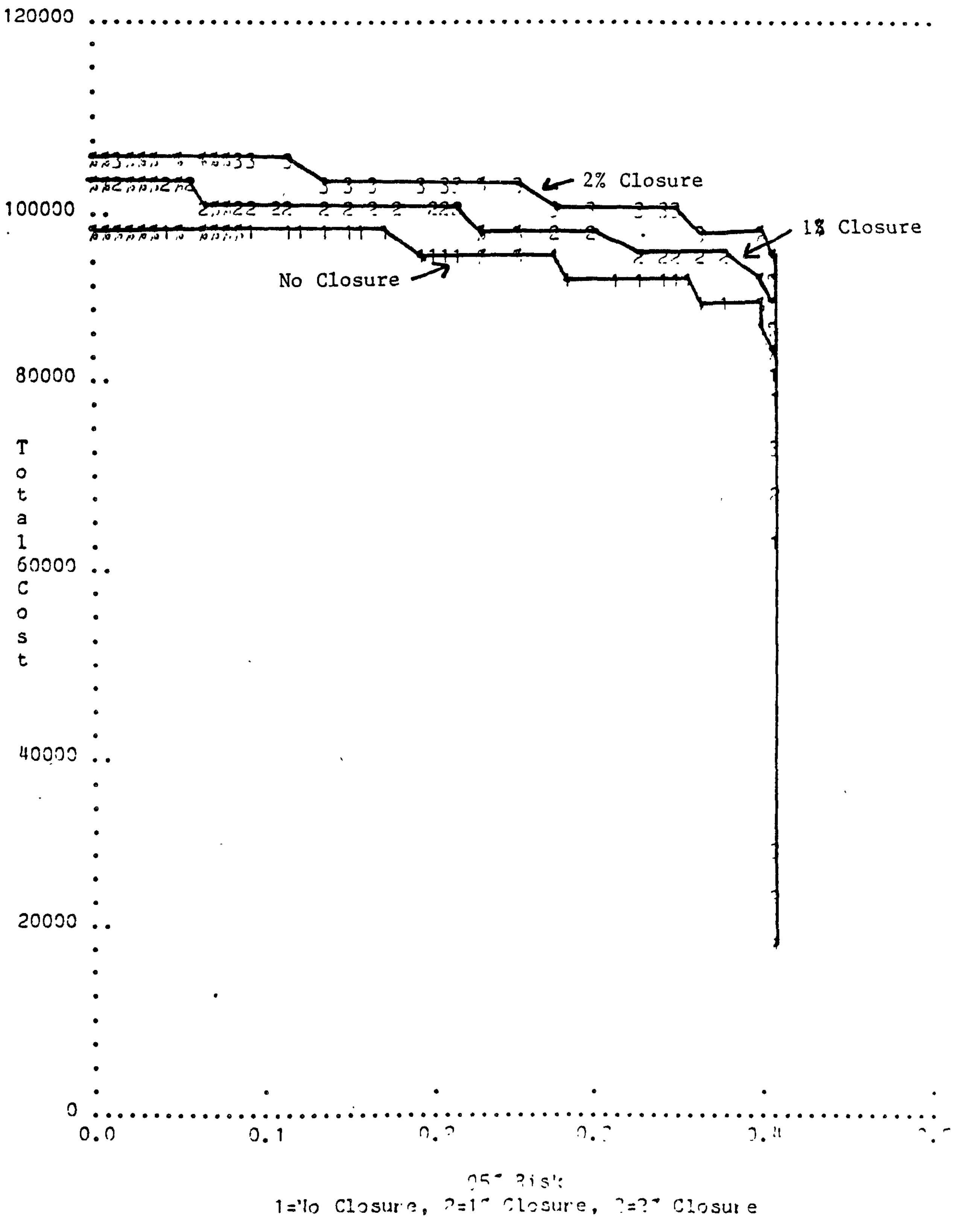


Figure 3

Construction vs. Closure: 99% Safety Margin Margin

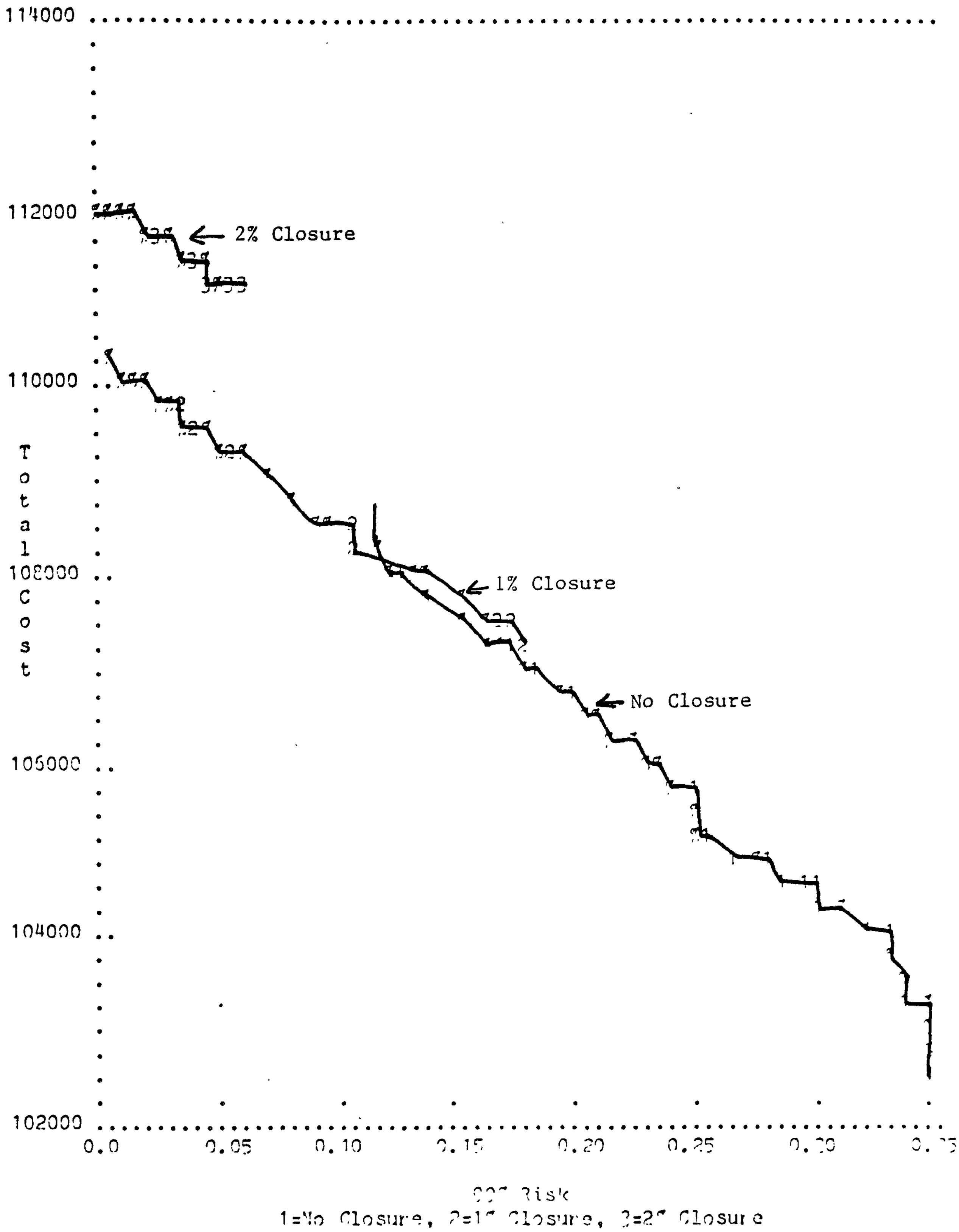
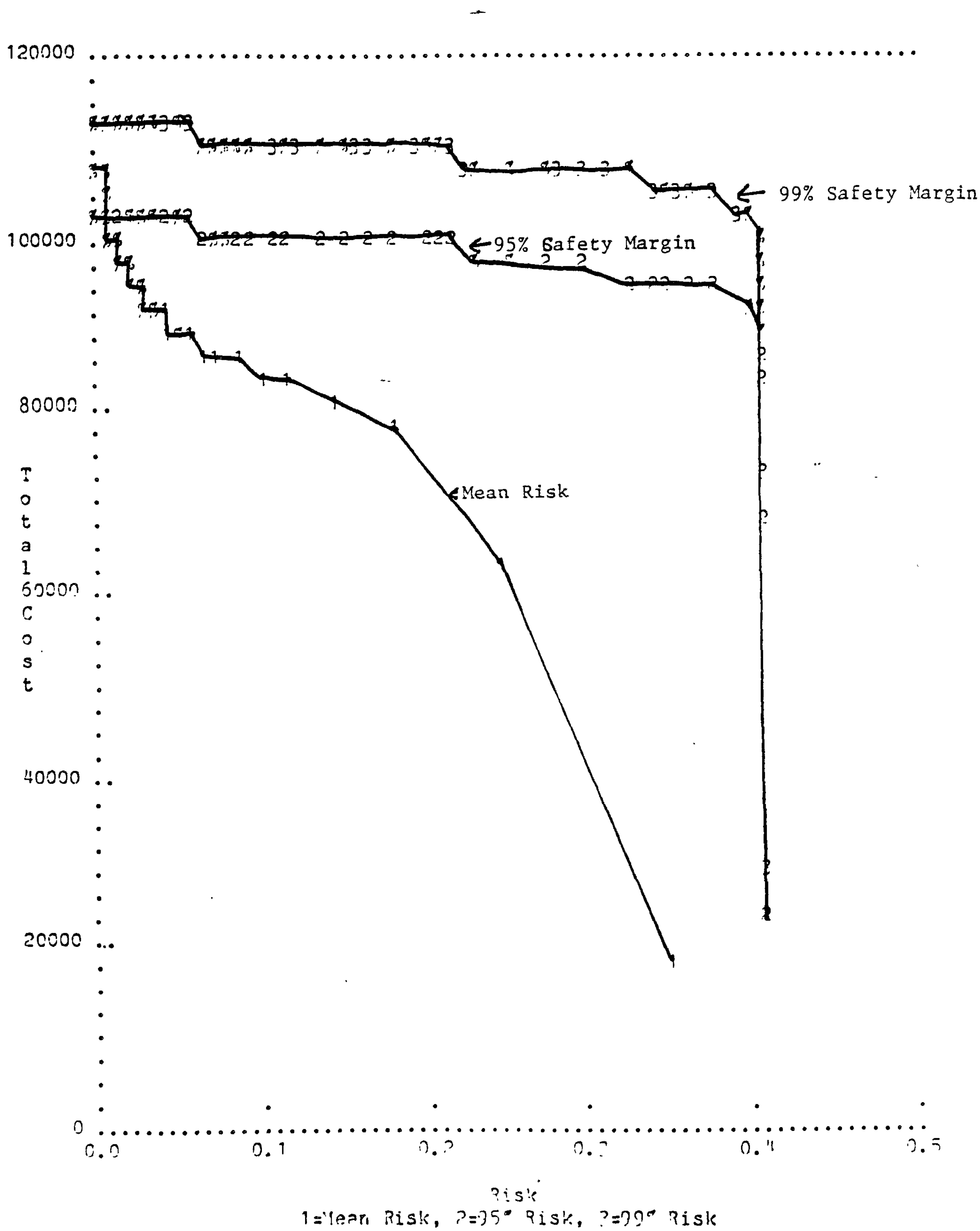


Figure 4

Impacts of Aversion to Uncertainty



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Mitigating Damages from Coastal Wetlands Development: Policy, Economics and Financing*

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INTRODUCTION

In the last three decades it has become increasingly clear that the wetlands are an integral component of the ecosystem of the coastal marine environment. The ascribed role of wetlands in the marine environment extends beyond such obvious functions as nursery and habitat to the contribution of wetlands to the nutrient and energy budgets of estuarine systems. Nevertheless, the scientific evidence on the relationship of wetlands to fish and wildlife populations, to shoreline erosion control, and to water quality is still being developed and remains somewhat contradictory [Nixon, others]. Furthermore, at this time, the ability to identify the contribution of specific wetlands parcels to the functioning of the marine ecosystem is limited [Oviatt and Nixon, others]. However, the general importance of wetlands systems to the natural productivity of the marine environment is seldom a matter of scientific debate.

During the same period that the scientific evidence on the ecosystem role of wetlands was accumulating, it was also evident that the nation's wetlands stock had been depleted by direct development on former wetlands (e.g. San Francisco Bay) and by indirect degradation from economic development activities [OTA]. A particularly useful illustration of indirect degradation is provided by the current condition of the Louisiana coastal marsh. For thousands of years the Mississippi River has meandered over the landscape creating a pattern of bank erosion and streambed deposition over its entire length. As a result the river has carried a heavy sediment load to its mouth along the Louisiana coast. As the flow rate slowed near the River's mouth, sediments were deposited and a delta was formed which developed into extensive marsh lands. The pattern of sediment delivery reached a dynamic equilibrium with forces of subsidence and shoreline erosion; over time vast acreages of marsh were maintained. At the same time a mix of vegetation types emerged and an extensive pattern of wetlands environments, ranging from saline to freshwater, developed.

Beginning in the last century, economic development began to intrude on this hydrologic process. Upstream tributary reservoirs were constructed and served as sediment traps; bank erosion was controlled. The result was a reduction in sediment delivery (NWS]. At the mouth of the river, navigation channel development constricted the meandering channel and funneled sediment and freshwater far offshore. At the same time, channels were cut through the marsh for pipelines to carry offshore gas and oil to on-shore processing facilities. These pipeline cuts affected the salinity regimes. Today as a result of development induced sediment starvation and changes in the salinity regime, Louisiana wetlands are disappearing at a rate far in excess of what would otherwise be the case. Subsidence and erosion are no longer being checked by sediment deposition and vegetation losses, and, as marsh changes from fresh to saline with salt water intrusion marsh soils are left unanchored by vegetation and subject to erosion [Watson, 1984].

Until recently state and federal policy encouraged "reclamation" of wetlands for commercial use. However, during the last three decades, in response to the changing views on the importance of wetlands, this policy position has been modified and, at all levels of government, programs have been adopted to manage the rate and location of wetlands alterations. However, because scientific study of the relationship of wetlands to the marine environment is a relatively new endeavor, the establishment of scientifically defensible wetlands management goals and strategies of wetlands management has been difficult. In addition, there have been cases where damages to coastal marsh have occurred and legal actions have been taken to secure compensation for such damages. In those instances where wetlands damage has occurred, the basis for determining how much, if any, compensation is required also has been difficult.

We will first describe the technical obstacles to measuring the economic value of a coastal marsh environment. The focus will be upon the absence of scientific understanding and data suitable for sound valuation efforts. Next we will describe the rapid evolution of the nation's wetlands management programs. We will argue that the resulting policy has developed without a clear statement of goals and without a cost effective management program. One effect of this condition is that there is an ambiguous basis for ascribing economic penalties for damages to coastal marsh environments.

With this as background, we will propose a framework for wetlands management and damage assessment. Specifically, a policy based upon setting targets for maintaining regional wetlands stocks will be described, and the establishment of a wetlands development fee and wetlands banks will be outlined. The problem of wetlands valuation will addressed in this management context, and a defense will be offered for using the cost of wetlands construction and rehabilitation as a basis for “value” estimation. The last section of the paper will illustrate the application of the management and damage assessment approach for the Louisiana coastal marsh.

Valuing and Managing the Undeveloped Coastal Marsh: The Standard Efficiency Framework

The valuation of a natural wetlands can not be made by inference from land market prices because the value of many of the service flows will not be appropriable by the land owner. Thus valuing a marsh parcel as a fixed asset, which will yield a flow of services, requires valuing the service vector of the marsh parcel in question and then inputing the values of the service to the marsh itself. The general framework for evaluation of natural wetland areas is discussed below in the context of Figure 1.

The framework for valuation of unaltered natural wetlands builds upon the recognition that a wetlands area can be viewed as a physical asset which functions as part of both a hydrologic and ecologic system. In turn this functioning gives rise to at least some of the following services: recreation, water supply, erosion control, and wildlife and fish habitat. The possible existence of these services means that an economic use-value for a wetlands area may exist if the one or more of its services is scarce. Figure 1 depicts a valuation sequence from wetlands area (Box I) to wetlands function (Box II), to wetlands service (Box III), to service use-value (Box IV) along linkages (a), (b) and (c). Wetlands values in Box IV are money equivalent measures of value in terms of economic surplus. Wetlands valuation requires a basis for quantifying each of these linkages.

Also depicted in Figure 1 is the possibility that substitutes may exist for a wetlands service. Substitutes for a wetlands area (Box V) include construction of new wetlands by whatever means are technically available to replace a wetlands lost to development. The constructed wetlands would be presumed to provide similar functions as the lost area. Purchase and presentation of wetlands which would otherwise be lost to development may also be considered a substitute for a natural wetlands area. Substitutes for wetlands functions (Box VI) would include, but not be limited to water control structures, artificial ground water recharge, erosion control structures, water-water treatment plant and fish hatcheries. These are possible substitutes which can give rise to a hydrologic or ecologic function that might equally be derived from the wetlands area. Substitutes

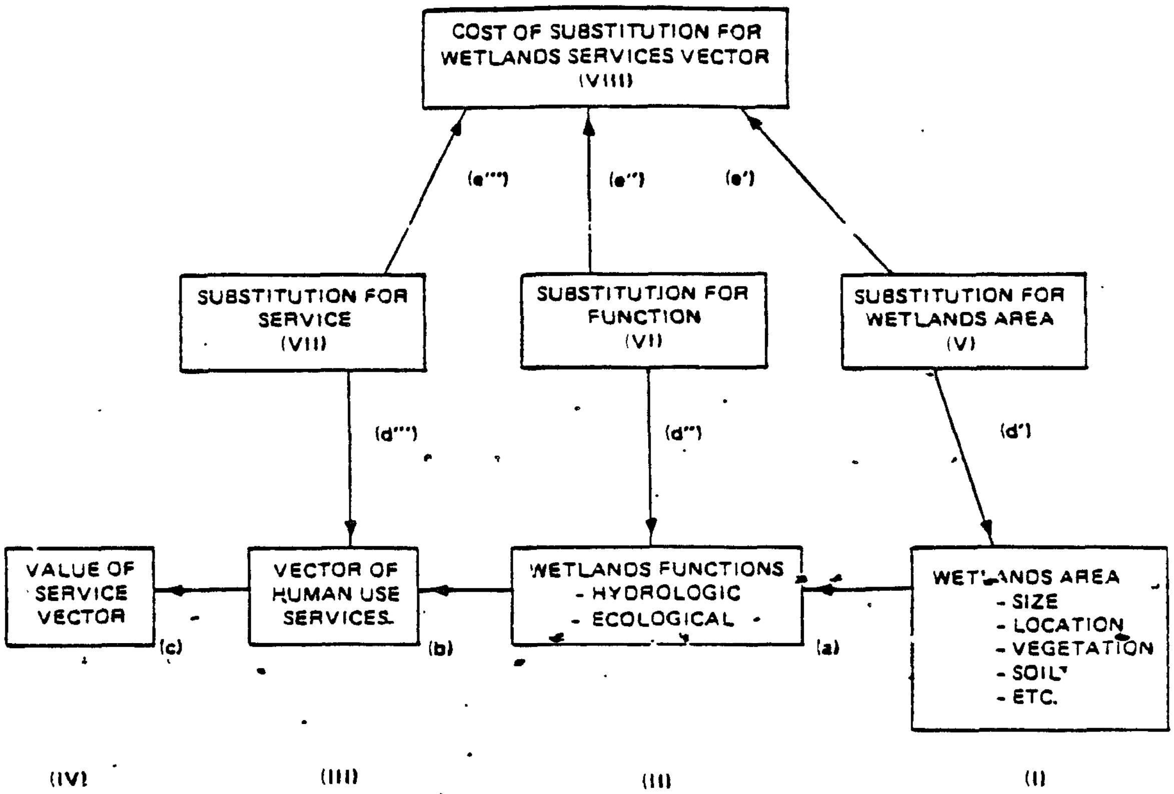

Figure 1. Determination of unaltered wetlands value.

for wetlands services (Box VIII) include all those possible actions that directly provide a service that can be provided by wetlands function. Such service substitutes would include but not be limited to, changes in location of economic activity to less flood prone or erosion prone land, provision of water from alternative sources or water conservation, harvest of commercial products from alternative areas and alternative recreational activities and sites.

The linkages (a), (b), (c), (d'), (d''), and (d''') suggest that the wetlands services available at any time and place will depend upon the mix of wetlands areas with area, as well as function and service substitutes. Linkages (e'), (e'') and (e''') suggest that the cost of wetlands substitution (Box VIII) depends upon the particular mix of actions taken from Boxes V, VI, and VII.

The economic approach to valuation of a natural wetlands area can be readily shown with Figure 1. The basis for establishment of wetlands area value is the "with and without" principle. Specifically, to the extent that a natural wetlands area is the only source of wetlands services, the foregone economic value of these services (Box IV) with the change in wetlands area, as compared to without the change, can be entirely attributed to the wetlands area change along arrows (c), (b) and (a).

Of particular importance is that the "with versus without" principle of valuation requires that valuation be specific to each area where wetlands' change is being considered. Site-specific analysis is implied for at least two reasons. First, a wetlands may provide a service but no use-value will be lost if the wetlands area is developed. For example, a wetlands that is recharging groundwater might have no water supply use-value if all water users in the area are dependent upon an upstream reservoir for water supply. Second, not all areas are equally capable of providing a wetlands service. Thus to observe that a service exists with the wetlands area does not logically permit the analyst to conclude that the service will cease to exist without the wetlands area. For example, a particular commercial fish harvest level which exists with a wetlands area may not change when the wetlands are developed, if changes in fish populations with the wetlands alteration are too small to affect harvest. Continuation of small reductions in wetlands acreage over time may result in loss of a service if the cumulative losses reduce wetlands acreage to below some threshold level needed to maintain the service level. At this point positive marginal values for each remaining acre will rise.

The “with versus without” argument can be described as the isolation of the value of a marginal acre of wetlands. Documentation that wetlands on average may provide a wide array of services does not serve to establish marginal values for specific land parcels. Furthermore the “with and without” principle requires that the measure of natural wetlands value must consider whether wetlands substitutes exist. If substitutes do exist, then the cost of employing substitutes to replace the wetlands service must be considered as an alternative measure of value. This is traced along arrows (d'), (d'', (d''') and (e'), (e''), (e'''). The value of a wetlands area will be the lesser of (1) the least cost combination of wetlands substitutes capable of providing the same services, (Box VIII) or (2) the direct measure of service value attributable to the area (Box IV). .This is so because people would not be willing to pay for a wetlands any more than the lesser of the value of the services it provides or the cost of replacing the services by employing a wetlands substitute.

It is, therefore, also the case that the cost of replacing the services is the upper limit on wetlands value. The standard criticism of using replacement cost is that it can easily overstate economic surplus measures, unless there is evidence that people would be willing to pay for the wetlands service if it were sold (presumably by a discriminating monopolist) at prices sufficient to cover the cost of the substitute. The fact that replacement cost is an upper bound on economic value measurement is not as commonly acknowledged.

When identifying the cost of a wetlands substitute, consideration should be given to the “least-cost” substitute. In finding the least-cost alternative combinations of service, function and area substitutes, combinations would be chosen and implemented in stages so that the wetlands service vector is replaced at lowest cost. This search for the the least-cost alternative is necessary since the presumption is that people would only be willing to pay an amount equal to the least-cost way of replacing the service if it were lost. However, identification of the least-cost alternative combination will require knowing that service, function and area substitutes could substitute for the wetland area being evaluated. In fact, the wetlands area, now or at some future date must be likely to provide the service flow. For example, purification of wastewater can only be a value for a wetlands if the area in fact receives and processes wastewater. If an area does not receive wastewater, then it would need to be demonstrated that ambient water quality in an area would decline without the specific wetlands parcel.

As an alternative to identifying the least cost combination of substitutes, the physical substitution of a wetlands area can be presumed to replace whatever services were flowing from that area without having actual knowledge of linkages in Figure 1. The wetlands assessment process needs to determine the structural features of the wetlands area to be valued as a basis for insuring substitution of ecological and hydrological function. Then, it can be presumed that the service vector of the substitute wetlands will be identical to the service vector of the area being valued. However, the service vector itself (Box II, in Figure 1) need not be known. Cost of a wetlands area substitute can be estimated by standard engineering cost estimation methods. These cost estimates would be a maximum measure of the value of the service vector (Box IV) and, hence, the wetlands area being replaced (Box I). This represents a maximum value measure because it will be equal to or greater than the value of the service vector as well as equal to or greater than the least cost combination of wetland substitutes.

It should be clear that accurate use valuation depends upon extensive wetlands function analysis. First, if natural wetlands services value is to be imputed to a wetlands area, linkages (a), (b), and (c) in Figure 1, it is necessary to identify services produced by a specific area and the level of service provision. Otherwise, the analyst may establish a potential natural wetlands services value (Box IV), for example for recreational fishing, but not be certain that the wetlands area being valued is providing that service. A second reason for needing function and service assessment analysis is to be able to specify whether substitutes are able to produce the same services as the wetlands area being valued -- linkages (e') (e'') (e''').

Within this context, what are the prospects for establishing the asset value of a natural wetlands acre? Practical limitations on the application of the valuation methods available for natural wetlands services are numerous. Such valuation requires detailed function and service assessment studies for specific wetlands areas in order to establish whether such areas are capable of producing a particular wetlands service. Since all wetlands are not of equal productivity, the needed detailed assessment analysis will be both tune consuming and costly. These assessment needs will stretch the ability of wetlands scientists to make scientifically defensible claims about the services of any given wetlands in the larger marine environment.

Oviatt *et al.* [ ] after reviewing the knowledge of ecosystems functions and services of coastal marsh in Rhode Island offer an assessment of the difficulties posed a measurement goal. They state:

> We conclude that the evaluation of individual coastal salt marshes in terms of their environmental contribution is not likely to lead to successful management strategy. (p. 210)

They reach this conclusion from a perspective which argues that individual wetlands areas do not exist in isolation from each other in terms of the hydrologic or ecological functioning. There is a synergistic effect among marsh areas as opposed to an additive-effect and, therefore, they conclude that "development of ecological rating systems for coastal wetlands must be viewed with considerable skepticism, for there is little reason to believe that such systems can be established on sound scientific ground." (p. 211)

When it is recognized the economic evaluation of a marsh assets will depend upon the scientific base of the ecosystem argument it is essential to recognize the concluding comments of Oviatt et al. on the scientific foundation for wetlands management

> It [wetlands management] should not be made on the basis of an elaborate rating scheme that is, in fact, built on a very shaky intellectual foundation. It is a great disservice to pretend to so much certainty when we are still far from knowing what is happening in the wetlands. (p. 211).

Application of economic valuation techniques in individual cases will require substantial analytical cost and time requirements, especially as more complex theoretical considerations must be made. These cost and time requirements mean that for numerous small wetlands areas, careful valuation of services will be infeasible. Unfortunately, general value estimates are not applicable to individual areas because of the site specific differences in productivity of wetlands areas and the site specific nature of the demand for wetlands services.

Despite these measurement difficulties, a perception persists among economists and others, that it is necessary to ascribe economic values to individual wetlands areas. Such a perception persists within the economics discipline because of the strong orientation to the economic efficiency perspective on resource allocation which requires analysis of marginal gains and costs over a continuing sequence of decisions. However, there are other opportunities for economic analysts to serve wetlands policy and management. The scope of this alternative economic work is discussed in detail after a brief review of current wetlands programs.

The Wetlands Policy Context

The nation has no explicit policy governing the protection of wetlands from development. Rather wetlands politics and programs are the product of a diffuse and evolutionary process that implies favoring wetlands protection over wetlands alterations. The rate of wetlands alteration has slowed substantially during the last two decades, but the losses each year remain substantial [OTA]. In part these losses are the result of a continuing effect from previous wetland destruction (such as that affecting the Louisiana coast), where there is not an explicit program for restoration. At the same time, because regulation proceeds on a case by case basis, wetlands regulation has been only a partially effective tool in a diffused and contradictory regulatory environment. In a recent speech, Milton Russell, Assistant Administrator for Policy, Planning and Evaluation at the EPA, summarized the situation by noting that "our current wetland policy is confused and often inadvertently destructive" (p. 8). Such a conclusion is consistent with the findings of other studies [OTA].

It is instructive to briefly review how the national policy developed and to note its fundamental differences with other environmental programs. In the late 1960's a series of court interpretations of the 1899 Rivers and Harbors Act required that the U.S. Army Corps of Engineers expand their review of dredge and fill permit applications to include not only obstructions to navigation, but also the effects of fill activities on wildlife habitat. This judicial action was intended to bring the Corps' permitting program into compliance with the requirements of the Fish and Wildlife Coordination Act of 1958 (FWCA). However, the FWCA required only that of habitat effects be considered, in decisionmaking; there was not a no mandate to protect habitat.

Questions which arose immediately were whether the limited jurisdiction of the Corps' permit program, to the navigable waters of the United States, included wetlands adjacent to all water bodies. Another matter in need of interpretation was whether the effects on habitat were to be only at the site or were to also include possible indirect effects of filling activity. The passage of the National Environmental Policy Act (NEPA) of 1969 served to expand the required review of permits to environmental concerns beyond wildlife habitat, if such a permit was deemed to be a “significant” federal action. However, NEPA, like the FWCA, only required that consideration be given to environmental impacts and carried no substantive statement of environmental goals.

Although legislative action to clarify the national policy to wetlands development would have been desirable, the actions of the Congress in the 1972 amendments to the Federal Water Pollution Control Act (FWPCA) did not clarify wetlands policy; yet it is Section 404 of the Act which is the basis for the existing federal wetlands program. In Section 404 the Congress expanded the Corps’ permit authority to require that permits be denied when permitted activities would adversely affect either navigation (under the 1899 Act) or water quality standards as established in compliance with the FWPCA of 1972. (There were some exceptions to the scope of this regulatory authority). Proponents of wetlands protection subsequently filed a series of court cases to argue that there was a demonstrable link between wetlands and adjacent water quality and that, therefore, it was the intent of the Congress in framing Section 404 that the Corps be responsible for review of proposed development in all wetlands. At the same time the NEPA process and the FWCA requirements remained in effect, and the conclusion often was made that an overall federal wetlands protection-strategy had been pieced together.

However, there was no concurrence among the federal agencies, among the states, or within the larger public that Section 404 was intended as a wetlands protection program. For example, the Congress left unaddressed issues of jurisdiction-- e.g. which wetlands are part of the navigable waters and whether indirect effects of filling were to be considered in permit review. To this date there remains a debate over whether it is only the direct water quality effects of wetlands filling which are covered by Section 404.

Wetlands protection efforts have been analogous to the manner in which wilderness and national park policy is made without any definition of standards to be achieved; in that regard it was unlike water quality management which is directed by stream and effluent standards. Although this discussion has focused upon the federal program, the problems in the states were analogous ones and legal indeterminancy and scientific uncertainty have resulted in a variety of different approaches to wetlands management. The result of the national process leading to the current wetlands policy setting is that most attention has been paid to regulatory tactics both by those who wish to restrict wetlands alternations and by those who feel some wetlands conversions should be accommodated. A discussion of the twists and turns of this process, usually in the courts, would be interesting but not germane. What has been missing is a clear statement of goals and standards for wetlands regulation to give a larger context to the regulatory framework.

Regulatory critics have argued (with justification) that, at times, Section 404 regulation has been unresponsive to cost considerations [RIA] and often inflexible even when those who wish to develop wetlands offered compensation [Tenneco]. Without a more complete statement of the purposes of the wetlands regulation it is unlikely that these objections to the program will be overcome.

In part the regulatory dilemma has been a product of the fundamentally weak scientific understanding of wetlands relationship to water quality and other environmental services. Denial of a wetlands development proposal requires the permit agency to demonstrate the negative affect such development would have on water quality. If the agency wished to consider other possible environmental values, there was an equally weak scientific basis for linking a particular wetlands permit to the larger environment. As Milton Russell noted in his recent talk, “It would be easier if we knew more. If the various wetlands ecologies were really understood, we could make intelligent defensible resource decisions.” (p. 9)

Directly analogous to the regulatory management problem has been the problem of assessing damages to wetlands. As awareness of the importance of natural environments has increased, there has been a number of court cases seeking compensation for damage to natural environments, including coastal wetlands. In one case, the defendants were the owners of the *S S Zoe Colocotroni,* a tramp oil tanker which ran aground off Puerto Rico in 1973. In order to refloat the ship, the captain jettisoned over 1.5 million gallons of crude oil. The oil came ashore in the Puerto Rican bay of Bahia Sucia where, despite cleanup efforts, there was substantial damage to benthic and intertidal organisms and mangroves. The Commonwealth of Puerto Rico and the Environmental Quality Board sued for damages (Commonwealth, 1980). The court, after hearing various competing views on the calculation of damages, awarded the plaintiffs $6.2 million. This damage was determined by estimating that there had been a decline of 4.6 million organisms per acre due to the oil spill; then, a price of 6 cents per organism charged by biological supply laboratories was used to compute an organism replacement cost of $5.5 million. The judge then added to this figure the costs of replanting 23 acres of mangroves and the cost of oil cleanup to derive the $6.2 million total. This damage award was appealed and the higher court struck down the computation of damages by use of prices from biological supply catalogs. The higher court concluded with respect to estimating damages that,

> To say the law on this question is unsettled is vastly to understate the situation... we....have ventured far into uncharted waters.. (and cannot)... anticipate where the journey will take us [Commonwealth, 1980, p. 46].

The case was remanded to the original court with the suggestion that the plaintiffs may wish to consider such items as alternative site restoration in computing money damages.

The *S. S. Zoe Colocotroni* case used the principle of replacement cost but the application was found inappropriate. In this case, the damages were measured by the cost of replacing one service (organisms) rather than the physical system that supported the organisms. It is extremely doubtful that society would demand the organisms, if available at $5.5 million, or that such a replacement was a least cost alternative. Indeed the higher court recognized these problems when they stated that awarding actual wetlands restoration costs is

> a far different matter from permitting the state to recover money damages for the loss of small, commercially valueless creatures which assertly would perish if returned to oil-soaked sands, yet probably would replenish themselves naturally if and when restoration - either artificial or natural - took place [Commonwealth, 1980].

Recently, another case focused, in part, on wetlands damage. In 1979, Louisiana placed a “first use tax” on natural gas which was pumped off-shore but which was “first” processed in Louisiana before being transported to out-of-state markets [Louisiana, 1979]. Louisiana claimed that the natural gas processing and transportation facilities had caused severe damage to the state’s coastal wetlands and that the tax proceeds were designated to restore and maintain the wetlands.

It was on this basis that Louisiana claimed that tax collections of $264 million per year were “fairly related” to wetlands damages. The states of Maryland, Illinois, Indiana, Massachusetts, Michigan, New York, Rhode Island, and Wisconsin sued to stop the tax from being implemented. As one argument in the case, the plaintiffs claimed the tax levy exceeded environmental damages.

While it was relatively easy to establish that some wetlands damage had occurred--particularly due to pipeline placement and right-of-way maintenance-- the question of assigning damages for establishing the tax was an important basis for the court challenge. On June 15, 1981, the U.S. Supreme Court struck down the Louisiana "first use" tax as unconstitutional and ordered all previously collected taxes refunded with interest; however, the decision was not based upon a consideration of whether the tax was fairly related to environmental damages. Therefore, the basic question of assigning damages remains an open one for future cases of this type.

Toward a Rationalized Wetlands Policy

Given the current state of wetlands science and national wetlands policy, what contribution can economic analysis make to an improvement? We have argued elsewhere that the current state of understanding of wetlands asset values, as distinct from individual service values, makes a case for a policy bias favoring wetlands preservation unless the costs of such a bias are unacceptably large in any specific instance [Batie and Mabbs-Zeno, Shabman, Batie and Mabbs-Zeno, Shabman and Bertelson]. We also would argue that a policy based upon a benefit-cost balancing test for wetlands permitting is technically impractical. Finally there is sufficient evidence that, despite the ambiguity of many aspects of wetlands policy, there has been a national shift from viewing wetlands as wastelands to viewing wetlands as national assets in need of protection. The shift in viewpoint is a reflection of fundamental realignments of the recognized implicit and explicit property claims to use of these landscapes. These realignments are away from solely private discretion in determining the fate of a land parcel to a sharing of that decision with regulatory authorities. The essence of these general arguments is that the economic policy question of interest is not whether wetlands should be preserved but rather over what the most efficient manner to achieve that goal. Milton Russell summarized his views on this argument by concluding

> responsible position, it seems to me, is to avoid casual, uncaring destruction. It is to raise the hurdle' over which those who want to convert or otherwise damage wetlands must jump. In time research may show that these hurdles can be lowered. But unless we set them high now, wetlands research will be of merely academic interest... (p. 10)

To frame an economic policy problem in this context requires that a wetlands goal be established and that institutional reforms be designed which are able to achieve that goal at least cost. This same perspective underlies applied environmental economists' arguments in support of effluent taxes and transferable pollution rights (TPR). There is a potential to transfer the logic of those proposals from their typical application to water and air quality to wetlands management.

In order to make the rationale for this transfer more clear, it is necess
for example, a effluent tax system is supposed to work.

In the effluent system the policy decision is divided into two
over time can cause the two parts to become interdependent
environmental goal is established for a particular area such as
is in turn translated into a maximum allowable waste discharge
environmental quality to remain within the stated goal. While economic
a part of the process which defines the goal, the benefit analysis is not expected to be a definite guide.

With an ambient goal established, the second aspect of the policy is to design an institution to allocate waste reduction requirements such that the marginal cost of waste withholding is equated across all waste dischargers. In the effluent tax system, the price is set administratively to achieve this marginal cost pricing rule. At the same time, the eflluent tax reflects the marginal cost of waste withholding for the last firm which chooses to pay the tax.

For purposes of this argument, it is worth considering the economic implications of this approach in more detail. Specifically, the effluent tax is equal to the marginal cost each party bears to maintain the ambient standard. Stated differently, the effluent tax is equal to the marginal cost of compensating for an increase in discharge at one point by a reduction at another in order to maintain the ambient standard; the effluent tax is equal to the cost of one party *replacing* anothers waste withholding effort. As long as the cost for treating the waste rather than forgoing output for changing production practices is the waste withholding cost, then a measurement of the marginal cost of waste treatment is a sound basis for setting the initial effluent fee. In any event, this is the recommended approach for setting an initial fee [Kneese and Bower]. In short, replacement cost is the accepted basis for a practical effluent tax scheme.

There are a number of practical reasons to build a wetlands policy on the replacement cost basis. In addition such an approach is analogous to the typical effluent tax proposals favored in applied environmental economics studies. In establishing a wetlands policy on a replacement cost foundation it must be established that wetlands are replaceable either by restoration efforts or by the creation of new wetlands. Indeed, the evidence is accumulating that wetlands construction and replacement is a technically achieveable practice that should be integrated into any wetlands management program. The feasibility of wetlands construction and rehabilitation has been a subject of recent research and practical experimentation [Dunston et al. 1975, Saucier 1978, Garbish 1978, Cole 1979]. Preliminary investigation of marsh creation in coastal areas suggest that constructed march areas offer many of the same services as natural marsh after one to three years [Garbish 1978, Jerome 1979, Newling 1981]. Ashe [1982] argues that restoration of previously altered areas may be a more feasible and suitable means of maintaining a wetlands base than new wetlands construction. Areas physically altered in the past are easily identified and may only require minor amounts of remedial engineering activity to restore wetlands functions and services.

The physical replacement or restoration of one wetlands area can be presumed to replace whatever services were flowing from another area which was developed, without having actual knowledge of linkage (a) in Figure 1. The biological assessment process need only determine the structural and hydrologic features of the wetlands area to be replaced as a basis for insuring physical replacement of those features. Then, it can be presumed that the service vector of the replacement wetlands (along arrow d) will be identical to the service vector of the replaced area (along arrow a). However, the service vector itself (Box II, in Figure 1) need not be known. Costs of wetlands replacement or restoration can be estimated from standard engineering cost estimation methods.

Wetlands management must be targeted to an eco-region [FWS] much as water quality management can only be pursued in a specific watershed context. However, the possibility of demonstrating conclusively the relationship of individual wetlands parcels to the larger eco-region is limited at best. As a result, wetland policy reform must begin by initiating a process of goal setting for maintenance of minimum wetlands stocks of various types [FWS] within defined eco-regions. In principle all wetlands may be replaceable, however, certain areas would no doubt be reserved from development (e.g. “wetlands wilderness areas”) in the setting of a wetlands policy. Given the historical loss rate of wetlands, and the continuing scientific uncertainty about their relationship to the larger ecosystem, a bias toward preservation of something close to the present wetlands stocks seems warranted. The role of economic benefit analysis in such a process is not clear, however, basic opportunity cost and marginal value principles can help structure the discussion and debate over goal setting. To quote Milton Russell once more,

> there is not enough known to do that [set a wetland stock goal], it strikes me that the only sensible policy is to start pushing harder on the brakes, wherever wetlands are threatened. (p. 10)

This recommendation to begin by goal setting is made neither casually or naively. The costs of developing the information base needed to inventory wetlands and mapping their location will be substantial. However, such an effort has been underway for a number of years within the Fish and Wildlife Service. In addition the maintenance goal will not be expected to be static over time, but rather would respond to new scientific information on wetlands and to recognition of the actual replacement costs of maintaining a goal (actual replacement costs are discussed in detail below). Such a dynamic approach to goal setting is of course descriptive of the type of trial and error process that always characterizes decision making where neither technical or value information is ever attainable [Wildavsky].

Wherever wetlands can be created and restored it is possible to permit actions which destroy wetlands as long as offsetting actions are taken to reestablish a wetlands at another location in order to maintain the wetlands stock. This is the exact logic behind using effluent fees and TPR systems to maintain an ambient environmental standard. The replacement action may be taken by the wetlands developer as part of the regulatory agreement which yields the permit. Replacement may be by purchase and preservation of wetlands which would otherwise be lost to development or by construction/rehabilitation of another wetlands areas. In fact, such mitigation requirements are occasionally made a condition of wetlands conversion permits. Recently, some large corporations have proposed that developers, or a group of developers, set aside wetlands areas and establish new areas that would serve as "wetlands banks". These credits for wetlands creation could be drawn upon in instances where wetlands were destroyed as part of their commercial development activities [Tenneco].

An extension of this concept is to allow the developer to make a money payment to the permitting agency and the agency would then use such money for wetlands replacement. The agency could collect wetlands conversion fees and, when revenues were sufficient, could initiate a wetlands construction/restoration project. Alternately the agency could construct wetlands and then collect fees to recover costs. Such a development fee would be set equal to the marginal cost of wetlands replacement. As will be noted below, the fee system may be especially attractive if there exist scale economies in wetlands construction which can only be realized by the management agency. Thus, the second aspect of a wetlands policy reform is to establish a system of fees based upon replacement cost of wetlands services in conjunction with a full mitigation requirement for all wetlands development proposals. As long as the entity causing the damage either provides in kind compensation or pays the fee, the proposed development should proceed. The regulatory problem, once goals are established, becomes one of insuring that the mitigation provisions are appropriate rather than one of trying to assess the benefits and costs of each individual permit prior to making a decision In this way, the proponents of the development face an opportunity cost based upon a specified and well articulated wetlands policy and they can choose to individually make the adjustments in their development plan which are most appropriate to their own situation.

This policy setting also provides a context for establishing charges for damages to coastal marsh. With the replacement cost policy in place and estimates of replacement cost fees set, there is a basis for charges to be assigned for assessing damages to coastal marsh. To the extent that it can be demonstrated that only a partial loss of wetlands services has been caused (e.g. short term chemical contamination), then the replacement cost fee would be an upper limit on the damage assessment. In addition the practical advantages to the replacement cost approach to damage assessment is that it provides a readily understandable basis for ascribing damages to an environmental alteration. The importance of having generally understandable measurement approaches is argued persuasively by Kimm *et. al.* (1981) in a discussion of benefit-cost analysis in the EPA regulatory process.

> (I)t is predictable that both (or all) sides will be able to then bring forward analyses that support their points of view. Each will then bring forward analysts, of equivalent credentials, who will then argue among themselves about the superiority of their particular analyses in a language all their own. Incomprehensible analysis may then drive out understandable argument [Kimm, *et. al.* 1981, p. 242].

An illustration of the measurement of replacement costs for the Louisiana coast follows in the last section of this paper.

There are three general techniques for the manmade creation of wetlands in Louisiana. These are: controlled diversion, uncontrolled diversion, and the controlled placement of dredge materials. All attempt to duplicate, in some sense, the natural processes that have continued for eons--with sediment laden rivers providing the material for wetlands creation. It is only recently that the construction of dikes, dams, and navigation channels have either trapped sediment before it reaches the mouth of the river or which funnelled the sediment past the delta area to the deep waters of the Gulf of Mexico.

Controlled diversions involve building a structure into the river levee to provide for the controlled release of water through the structure into an area deemed suitable for wetlands. The controlled diversion could be by gravity flow structure, siphons, or pumping stations [Watson, 1984]. The diversion of the flow of the river to restore and to create wetlands has been considered and/or accomplished by the Corps of Engineers and the State of Louisiana [Gagliano and van Beek, 1976; Watson, 1984]. For example, the Violet Siphon Structure was completed in December 1979 in Louisiana's St. Bernard Parish. It consists of two 50 inch diameter pipes which divert a maximum of 250 cfs of Mississippi flow into wetlands lying behind the Mississippi River and Lake Borge [Coastal Environments, 1980]. This flow diversion carries sediment which nourishes existing wetlands and builds new wetlands.

Uncontrolled diversions are similar except that the breach in the levee does not include a control structure. Rather, an artificial crevasse simply diverts some of the river. Since uncontrolled diversion can mean major amounts of river flow through the crevasse during flooding, this technique is usually used well into the delta area, below any population centers.

The third technique used for wetlands construction is the placement of dredged material. Material dredged from navigation channels or sediment rich areas can be pumped into shallow water to create wetlands. Use of dredge material also assists in what otherwise would be a disposal problem, since over 60 million cubic yards of sediment are excavated annually in the Army Corps of Engineers New Orleans maintenance dredging program [Watson, 1984]. Furthermore, dredge disposal allows for increased flexibility in placement of the wetlands. Whereas, diversions limit wetlands creation to near the river channel, dredge material can be barged or siphoned to alternative areas. Landin estimated that it takes approximately 1613 cy/ft/ac of unconsolidated dredge material to build an acre of wetlands; in Louisiana, the material is placed so that, when it consolidates, the wetlands will be at the intertidal level [Landin, 1986]. Since 1970, over 15,000 acres of wetlands have been created from dredged material in Southern Louisiana. Most of this has been at Southwest Pass and the delta of the Mississippi. Over 4000 acres has also been built using dredged material in the Atchafalaya Basin and other parts fo the Louisiana Coast [Landin, 1986].

The placement of dredge material and the diversion of sediment bearing waters to build wetlands can be considered as a replacement for wetlands destroyed by economic development activities. Therefore, the costs of constructing new wetlands can be used to develop a wetlands management program within an eco-region such as the Louisiana Gulf Coast.

However, computation of the cost of a man-made wetlands area (i.e. a natural wetlands substitute) requires more than simply reporting the engineering cost estimates. This is the case because constructed wetlands may require the passage of time before they can provide the same service level as the natural wetlands area. Landin [1986], for example, estimates that it may require 3 to 5 years for a site in south Louisiana to appear as a natural wetlands; soil profiles and root biomass do not equal that of a natural wetlands until closer to 10 years. However, Landin [1986] further notes that once a wetlands has stabilized--anywhere from 3 to 10 years--there appears to be no functional difference between a manmade and a natural wetlands. Also, engineering costs may include a high initial capital cost component followed by low annual costs over time. Thus, the computation of costs require appropriate adjustments to account for time dimensions of replacement activities. Therefore replacement cost analysis requires the following information for valuing natural wetlands areas:

- What are the recurring annual costs (C) for the wetlands area substitute?
- What is the time period before the substitute can provide a service flow similar to the natural area?

Because the time paths of costs and wetlands service replacement may differ, the analytical problem is to determine the costs replacement. The determination of R, annualized replacement cost, by estimating the value of R which just equals engineering costs incurred for the years during which replacement actually is accomplished but which includes the time value of money. R can be solved for in the following expression. (Note that R is expressed in annual equivalent terms.)

$$0 = \left[\sum_{t=n}^{T} \frac{R_t}{(1+i)^t} \right] - \left[\sum_{t=1}^{n} K_t(1+i)^t + \sum_{t=1}^{T} \frac{C_t}{(1+i)^t} \right] \qquad (E1)$$

where

T = time period of analysis

t = year

n = year in which substitute provides wetlands service

R= equivalent annual replacement cost of service flow from created wetlands area

i = discount rate

K = costs incurred before substitute begins providing wetlands service (primarily capital costs)

C= recurring annual costs to keep wetlands substitute operating and maintained

In this equation, the present value of the annualized replacement cost of wetland services is shown as the expression in the first set of brackets. The second set of brackets is the actual expenditures for the wetlands substitute. Solving the equation for R yields the equivalent annual replacement cost of the wetlands area.

The general formula expressed above assumes that there are no transitional phases of wetlands development that yield valued services, however, the formula can be easily amended to include such transitional periods of construction.

Case Study-Controlled Diversion

Published technical studies of the potential for wetlands creation in Louisiana provided data with which to estimate the annual equivalent replacement cost. Table 1 describes 7 sites on the Mississippi River where controlled diversions have been studied. At each site, it was estimated that 320 acres of wetlands per year for 7 years could be created by controlled diversion of 25,000 cfs of river flow; construction costs for each site are displayed in column 2. Construction is expected to take two years to complete. Operation, maintenance, and replacement costs were estimated to be 4 percent of construction cost per year. The time horizon for the analysis is 50 years although wetlands creation ceases at the end of the seventh year. When the formula above is modified to account for the transitional growth of wetlands, the annual equivalent replacement wetlands can be determined for the 7 sites. These costs are displayed in columns (3) and (4). The difference in the estimates arise from the choice of discount rate: 10 or 5 percent.

Case Study- Uncontrolled Division

Uncontrolled diversions in Louisiana involve breaching a mainline levee with cuts varying from 50 to 200 feet wide. [Watson, 1984] These cuts imitate the natural process of river crevasse and overflow. Sediments from crevasses are deposited in low lying areas in deltaic splays. As the delta matures the rate of wetlands created decrcascs'over time.

Table 2 displays areas where potential wetlands creation with uncontrolled sediment diversions have bee studied. Over the first 2 years, a total of 420 acres of wetlands would be created at the 5 sites; however, over 50 years only 5200 will be created. The cuts have to be reopened and extended or relocated every 2 to 3 years to maintain the rate of wetlands rate building [Watson, 1984].

Assuming only one cut was made per site and was not reopened or relocated later, that wetlands building was such that, after 50 years, a total of approximately 5200 acres were created, that there were no maintenance costs after the second year, and that dredging costs are $2.13 per cubic yard, then the estimated annual replacement cost, R, vanes from $2.33 per acre to $54.21 per acre at a 10 percent discount rate and from to $1.26 per acre to $29.44 per acre at a 5 percent discount rate (see Table 2).

Table I
Wetlands Creation With Proposed 25,000 CFS
Controlled Sediment Diversions*

Site Location	(1) Wetlands Created acres/year	(2) Total Construction cost ($000)	(3) Annualized Replacement Cost Per Acre of Wetlands (10% discount rate) ($)	(4) Annualized Replacement Cost Per Acre of Wetlands (5% discount rate) ($)
Site 1 Mississippi River East Bank-Mi 34.9	320	$14,019	1244.51	724.37
Site 2 Mississippi River West Bank-Mi. 31.3	320	$27,446	2436.46	1418.14
Site 3 Mississippi River East Bank-Mi. 20.0	320	$16,307	1447.62	842.59
Site 4 Mississippi River West Bank-Mi. 16.4	320	$30,781	2732.52	1590.46
Site 5 Mississippi River East Bank-Mi.	320	$17,914	1590.28	925.62
Site 6 Mississippi River West Bank-Mi. 9.5	320	$19,275	1711.10	995.94
Site 7 Mississippi River West Bank-Mi. 6.0	320	$20,176	1791.08	1042.50
Average Sites 1-7	320	$20,845	1850.51	1077.09

* Modified from Watson, [1984]

Table 2
Wetlands Creations With
Uncontrolled Sediment Diversions*

Site Location	(1) Wetlands Created acres/year	(2) Total Construction cost	(3) Annualized Replacement Cost Per Acre of Wetlands (10% discount rate) ($)	(4) Annualized Replacement Cost Per Acre of Wetlands (5% discount rate) ($)
Site 1 Octave Pass, South Bank	3.7	115,020	54.20	29.44
Site 2 Raphael Pass, North Bank	3.7	57,510	29.10	14.72
Site 3 Pass a Loutre, North Bank	2.5	119,300	8.43	4.58
Site 4 Pass a Loutre, South Bank	50	60,000	2.33	1.26
Site 5 South Pass, West Bank	22.5	142,740	11.21	6.09
Average Site 1-7	21	75,933	21.05	11.22

* Modified from Watson, (1984)

Case Study - Controlled Placement of Dredge Material

The use of dredge material for wetland creation is a favored technique in Louisiana because it is lower cost than other techniques and because it provides an opportunity to use dredge material in a beneficial way. Since the erosion and subsidence of the Louisiana coast is so evident, there is little public opposition to dredge material placement [Landin, 1986].

Landin [1986] estimates that the average amount of dredge material per acre in Louisiana was 4800 cy. In South Louisiana dredging plus dredge material placement averages $1.23/cy. Clearly these costs will be highly sensitive to transport costs. Assuming it takes 3 years to create 2240 acres, that these acres begin to function in the third year, and that there is no maintenance costs after the third year, the value of R annualized replacement cost of services would be $658.89 per acre at a 10 percent discount rate or $345.03 per acre at a 5 percent discount rate.

Summary

During the past decade federal and state government regulation has sought to reduce the rate of coastal wetlands development. However, it is uncertain whether these programs can secure mitigation of damages caused by future development. Revision of coastal wetlands management programs is needed and should involve several actions. First, for regions of concern, wetlands acreage targets would be set; no such targets now guide wetlands regulation programs. As a part of this effort certain areas or types of wetlands might be protected from development. Then, in areas not protected, development would be permitted if the developer paid a fixed wetlands development fee which was set in relation to cost of wetlands replacement. The realized development tax revenues would be used to finance a region-wide investment program which can realize scale economies in replacing most wetlands lost to development. This approach to wetlands management offers more assurance that coastal wetlands damage will be compensated, provides for a more certain regulatory environment for coastal development planning, and provides a policy relevant context for coastal wetlands damage assessment. The case illustration provided for the Louisiana area indicates that replacement costs can be readily computed for various wetlands creation techniques.

The estimated annualized replacement costs was least expensive for uncontrolled sediment diversions (assuming no maintenance costs after 2 years and a gradual building of 5200 acres after 50 years as a result of five different levee "cuts"). The average annual replacement cost per acre when valued at a 5 percent discount rate was $11.22. However, the rate of wetlands created by uncontrolled diversion is slow. After an initial creation of 420 acres over 2 years from the 5 sites, further wetlands growth averages only 5 percent per year.

Dredging material placement, in contrast, would result in annualized replacement costs (R) of $345 per acre, (at a 5 percent discount rate). This R value is, however, very sensitive to transport costs. Nevertheless, the rate and amount of wetlands created are controllable by the rate and amount of dredge material placement.

Controlled diversions were comparably very expensive. Annualized replacement costs averaged $1077 per acre (5 percent discount rate).

These annualized replacement costs can be thought of as the amount of annual fee that would be collected from a permitee if the fee were based on wetlands replacement cost. While the higher annualized replacement costs from controlled diversions, may seem high, consider that, in the first use tax case in Louisiana, state officials were collecting $264 million a year as "fairly related" compensation to wetlands damaged. Use of the expensive controlled diversions to replace the 10,000 acres of damaged wetlands would result in average annual payments from oil and gas producers of only $10.7 million--not overly large considering the returns to be made from oil and gas field developments and much less than $264 million. Use of uncontrolled sediment diversion for wetlands replacement, in contrast would imply average annual payments of only $112,200.

REFERENCES

[Will be provided with later draft]

MEASURING THE ECONOMIC DAMAGES ASSOCIATED WITH TERRESTRIAL POLLUTION OF MARINE ECOSYSTEMS

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PRELIMINARY DRAFT

INTRODUCTION

Marine resources are very often perceived as independent of events that occur on land, but in many instances, particularly for estuarine ecosystems this is not the case. Terrestrial activities, both natural and anthropogenic have critical implications for marine organisms. For example, the amount of rainfall far to the inland will partially determine the salinity levels of an embayment which will have a direct impact on the size of oyster populations. Of course, anthropogenic activity is going to be more important from a policy perspective. Examples of these activities include the conversion of wetlands, which serve as important nutrient sources for marine organisms and quite often provide nursery and spawning habitat, the terrestrial application of agricultural chemicals, which may reduce aquatic vegetation which will lead to reductions in important fish populations, and diversion of river water for industrial, agricultural or municipal purposes, which may have effects on salinity and nutrient levels of the estuary.

As the above examples suggest, many of the externalities associated with terrestrial activities do not have a direct impact on marine resources or marine environmental quality parameters which are a direct source of utility for man. For example, over a very large range of salinity levels, changes in the salinity of an estuary are not even perceptable to the typical person, even those swimming in the estuary. Rather, changes in social welfare will arise as the effects of the environmental change work their way through the food web or affect natural processes (such as nutrient cycles), having both direct and indirect effects on organisms or environmental quality parameters from which man derives value. In short, the terrestrial activitites often affect environmental goods which are inputs in an ecological production process for what might be regarded as environmental outputs. These environmental outputs are those environmental resources which are utilized by man to produce utility, either by directly consuming the resource (consumption need not be depleting) or by using the environmental good as an input in producing an economic output. This is admittedly an anthropocentric approach, but one which is consistent with the way in which allocation decisions are made concerning private resources.

For example, assume that policy makers are concerned about the soil erosion from agriculture and its subsequent effects on the population of a marine species that is either commercially or recreationally harvested. Further assume that the fish spawns in an estuarine environment and spends portions of its adult life in both the estuary and the high seas. The estuarine turbidity caused by the inland soil erosion might impact the fish species in many different ways. For example, the suspended sediments might settle out on top of fish eggs, which will deprive them of oxygen. In addition, the turbidity may interfere with the photosynthesis of many different plant species, which will have repercussions throughout the food web, but will ultimately deplete the availability of food for the fish species. Additionally, it could adversely affect the reproductive ability of organisms further down the food chain, or affect their ability to respire, which would ultimately affect the abundance of the fish which is commercially or recreationally exploited.

After even a cursory examination, it becomes apparent that in order to calculate the welfare losses associated with a terrestrial pollutant such as sediment from increased soil erosion, much information must be gathered. First, one must be able to specify the fashion in which inland soil erosion affects the turbidity of the estuary. One then needs to illustrate how the change in turbidity level affects the fish population through all the direct and indirect avenues. In general, this will require a complicated model of the ecosystem which would be commensurate with large scale macroeconomic or regional economic models in terms of their data requirements. Often these data are not available and alternative procedures must be sought. In addition, an economic model of the commercial or recreational exploitation of the fish species and the interaction of exploitation with natural ecological processes must be formulated.

This study suggests methodologies for valuing the damages which terrestrial activitites generate in aquatic ecosystems. While the discussion of these methodologies and their potential application in the valuation process is the primary focus of this paper, the paper has an additional goal. This goal is to expose natural scientists to the types of models and the kinds of analysis which economists can contribute to the resource managemnt process. This is very important, because, as the paper will illustrate, economists can not operate in this area without input from natural scientists. The nature of the problem implies that at a minimum, economists will need to rely on the counterparts in the sciences to provide critical data on resource populations and other environmental variables. More ideally, economists would work jointly with natural scientists, integrating detailed ecological models with economic models. In the former case, economists must make their counterparts in the natural sciences aware of the type and structure of the data they need, because what is appropriate for ecologic analysis may not be appropriate for economic analysis. In particular, natural scientists who can test their hypothesis by replication across many different field sites may not be collecting data in a sufficient time series to conduct economic analysis. Since the different field sites are likely to span only one market, the economist also will not be able to perform cross-sectional analysis. The data needs will be discussed more fully as the paper proceeds.

The concepts developed in the paper are illustrated by examining the impact of certain terrestrial activities on the productivity of the Appalachicola Estuary in Northwest Florida. Of particular interest is the ability to capture some of these ecological interactions in the absence of a detailed ecological model. The empirical section of the paper will implement some of the methodologies which are suggested in the conceptual part of the paper. The problem which is used to illustrate the methodology is that of measuring the estuarine damages associated with increased upstream water uptake by the growing city of Atlanta, as the estuarine oyster populations are sensitive to the freshwater input from the Appalachicola River.

WELFARE LOSSES ASSOCIATED WITH POPULATION CHANGES

It is clear that in order to be able to estimate the change in welfare associated with a change in the level of a particular terrestrial pollutant, one must be able to specify how the pollutant affects the levels of marine resources which are inputs into various types of economic activity. For expositional simplicity, it will be assumed that the marine resource is a fish species which is commercially but not recreationally exploited. The extension to recreationally exploited species and to environmental resources (both animate and inanimate) which are inputs to other types of economic activity is quite straightforward, although data might be even more difficult to obtain than the Fishery case.

Figure 1 contains a simplified representation of the relationships which must be known before the welfare losses are determined. For the time being; steps (A) through (F) will be ignored, and the discussion will proceed as if these effects are known. The emphasis of the discussion will be placed on how changes in the fish population generate changes in the net social benefits associated with fishing activity. Obviously, a reduced population of fish will generate an upward shift in the supply or marginal cost function. Is is important, however, to distinguish between changes in population generated by the environmental change, and changes in population due to fishing activity (including the response of fishing activity to the new environmental conditions). The mechanism for doing this is the equilibrium growth function[1], which shows the natural growth (F(X)) to a given stock (X) per unit time. In the absence of exploitation, natural growth will equal the net additions to the stock (dX/dt). Both the net additions (and natural growth) and the stock itself are traditionally measured in biomass units rather than in the terms of numbers of individual organisms. Figure 2 shows the equilibrium growth function which is essentially a measure of the productivity of the stock. Note that there are two natural equilibrium levels of the stock, one at a zero level of population, and the other at K, which is the maximum possible level of the population or the carrying capacity of the environment. The growth function can also be interpreted as an equilibrium catch function since population will remain unchanged if catch is equal to F(X) for a given population level. It is this interpretation as an equilibrium catch equation which allows for the separation of the effects of fishing activity and environmental change on the fish population.

One more step must be taken before this uncoupling of the population effects can be made, and that is to forge a relationship between the growth function (equilibrium catch equation) and the supply function. This is done in Figure 3. Actually, the supply function should not be viewed as a single function, as there will be a different supply function for each level of the fish population. Since each supply function is defined for a specific population level, and the growth function gives the equilibrium catch level defined for each population level, the point on each supply fucntion that is characterized by biological equilibrium (population remains constant) can be identified. LL represents the locus of these points of biological equilibria, which is discussed in more detail in Kahn and Kemp. The level of catch which is characterized by both biological and economic equilibrium is given by the intersection of the demand function, the supply function and the locus of biological equilibria. Net benefits associated with the activity are represented by the consumers' and producers' surplus of area ABC.

A harmful environmental change will shift the growth function downward and to the left (as in Figure 4) and therefore shift the locus of biological equilibira inward as in Figure 5. The loss in social benefits associated with the harmful environmental change is given by the change in consumers' plus producers' surplus, or area EDBC in Figure 5. Note that in the context of this model, one can distinguish between a change in population attributable to environmental changes (a shift of the locus of biological equilibria) and a change in population generated by changes in the level of fishing activity (a movement along the locus of biological equilibria).

The locus of biological equilibria has an appearance similar to the long-run backward bending supply curves discussed by both Copes and Clark. For the purposes of valuing environmental resources it is preferable not to think of supply as a long run equilibrium locus, since the identification of social benefits such as producers surplus becomes problematic if the supply curve is not truly a supply curve but an equilibrium locus. Each of the supply curves in Figure 3 is capable of identifying producers surplus for the state of the world (level of fish population) associated with that supply curve. This is precisely what one wants in a supply curve as the producers' surplus will adequately reflect the inputs' opportunity costs in the current state of the world. This difference in interpretation of the supply function appears at first to be rather trivial, but the distinction is very important, especially if one plans to empirically estimate the model. To differentiate the model proposed in this paper from that of the long-run average cost supply model of Copes, this model will be called the contemporaneous supply model.

With its emphasis on the contemporaneously defined supply curve, rather than the long-run supply curve, this study (along with the previous study by Kahn and Kemp which introduced this type of analysis) represents a significant departure from conventional fishery economics. The traditional based yield-effort types of models are quite convenient for deriving the optimal harvest and stock time paths, as well as the maximized fishery rent. However, they are much less well suited for examining benefits other than fisheries rent, such as consumers' and producers' surplus. Of course, if both the marginal cost curve (holding population constant) and the demand curve both approach the horizontal, the contemporaneous supply model loses some of its advantages.[2]

THE IMPACT OF ENVIRONMENTAL CHANGE ON THE GROWTH FUNCTION

As stated above, a harmful environmental change will shift the growth function downward, but the mechanism by which this downward shift occurs has not been discussed. For simplicity, it is assumed that the growth function can be well approximated by the logistic function suggested by Shaefer. This function is presented as Equation 1, where K is the carrying capacity of the environment, r is the intrinsic growth rate and X is the population of the organism.

$$F(X) = rX\left[1 - \frac{X}{K}\right] \quad (1)$$

A harmful environmental change can impact on the growth function in two ways, either by changing r, the intrinsic growth rate, or K, the carrying capacity. For example, the massive infusion of DDT into the environment in the 1950s and 1950s probably diminished r for birds of prey such as eagles and ospreys, but not K. The ability of the environment to support eagles and ospreys (K) was probably not significantly diminshed by DDT (although it may have been by other factors, such as loss of habitat). However, the presence of DDT in the raptors' bodies is believed to have interfered with their reproductive ability. This would serve to decrease r since the intrinsic growth rate is the maximum proportionate growth rate or the proportionate growth rate when the population is very small. Even though there may be abundant food and habitat, F(X) will fall. Alternatively, an environmental disturbance such as the destruction of habitat or pollution which reduces fish populations upon which the raptors prey will serve to reduce K which will in turn reduce F(X).

The effects of reducing r and K are not completly symmetric. For an unexploited population, a decrease in r will diminish F(X) but not lower the long-run equilibrium value of X. This can be seen by looking at equation (2) which is the solution to the differential equation (1) (when the population is unexploited F(X) = dX/dt). On the other hand, a decrease in K will lower both F(X) and the maximum population which X can obtain. Of course, in the presence of exploitation, the reduction in r will imply either a reduced population if effort is held constant or a reduced level of exploitation if population is to remain constant. The corresponding solution for the differential equation dX/dt when fishing effort (E) occurs is found in equation (3).[3]

$$X(t) = \frac{K}{1 + \left[\frac{K - X(t_0)}{X(t_0)}\right] e^{-rt}} \qquad (2)$$

$$\lim_{t \to \infty} X(t) = K(1 - E/r) \qquad (3)$$

COMPUTATION OF WELFARE LOSS IN THE CONTEXT OF AN ECOSYSTEM MODEL

For the time being, the ability to estimate an ecosystem model will be taken as given. An ecosystem model can be defined as a model which specifies (as a system of differential equations) the flows of mass or energy within the ecosystem, subject to the constraints of the system. Such constraints are both external to the organisms in the ecosystem (salinity levels, the availability of dissolved oxygen, nutrients, etc.) and internal (metabolic rates, reproductive potential, etc.). Although a discussion of the issues involved in estimation of such a model is beyond the scope of this paper, it is possible to estimate such a model using a combination of field and laboratory data. This process would be analogous to estimating cost functions based on engineering data.

If the parameters of the model can be estimated, the ecosystem model can be solved for given values of the exogenous variables. The solution will consist of equilibrium population levels of the various organisms, which can be used to derive the carrying capacity of the species of interest. If the system is unexploited, the carrying capacity is the equilibrium level. If the system is exploited, (and the level of exploitation known) the carrying capacity can be identified. Hence, it is possible to forge a link between the the exogenous environmental quality variables and the carrying capacity of the species of interest.

This link provides the ability to identify the marine welfare losses stemming from terrrestrial pollution, provided the demand and supply functions (or inverse demand and marginal cost functions) are known for the economic output for which the marine species is an input. Inverse demand and marginal cost functions can be specified as in equations (4) and (5), respectively. Π represents the inverse demand for catch, MC represents the marginal cost of catch, C represents the catch (in biomass terms), ρ_i represents a vector of input prices, p_s represents a vector of prices of subsitute commodities, and S represents a vector of socioeconomic variables. Equation (6) contains the natural growth function of equation (1) interpreted as an equilibrium catch equation.

$$MC = MC(C, \rho_i, X) \qquad (4)$$

$$\Pi = \Pi(C, \rho_{s,} S) \qquad (5)$$

$$C = rX - (r/K)X^2 \qquad (6)$$

Given these functions, the economic equilibrium condition that $MC=\Pi$ and the ecosystem model it is possible to estimate the changes in net social benefits associated with an environmental change. The ecosystem model may be solved assuming a baseline environment to yield baseline values of K and r, which can be plugged into the equilibrium catch equation. The right hand sides of equations (4) and (S) are then set equal to each other (fulfilling the equilibrium condition) and the right hand side of equation (6) substituted into the resulting equation. This equation will be a quadratic equation in X, for which the solution is the equilibrium population level (X^*). C^* can be determined by solving equation (6) for the catch level, which is an equilibrium catch in both the biological and economic sense. The net benefits associated with this commercial fishing activity can then be calculated according to equation (7).

$$\text{net benefits} = \int_0^{C^*} [\Pi - MC]\,dC \qquad (7)$$

The welfare losses associated with the environmental change can then be calculated by solving the ecosystem model with the new levels of the environemental parameters. This will return new values of r and K, which can again be used to solve the system of equations (4)-(6). The new values of C^* and X^* are then used in re-evaluating equation (7). The difference between the pre-change level of net benefits and the post-change level of net benefits represents the welfare change associated with the environmental change. Note that changes in environmental variables will alter net benefits through two avenues. Changes in the environmental variables will change the equilibrium population and thus shift MC, and it will also affect the level of C^*.

THE MEASUREMENT OF WELFARE CHANGE IN THE ABSENCE OF AN ECOSYSTEM MODEL

The above discussed model is predicated upon the existence of a comprehensive ecosystem model. If this model has in fact been estimated for the ecosystem of interest, the calculation of the welfare loss associated with environmental change is relatively straightfoward. Unfortunately, such an ecosystem model will not be available in most cases. The data requirements for this type of model are such that an extensive and expensive data collection effort must be made over a relatively long period of time. Very often policy decisions will have to be made in a more timely fashion, without the luxury of first estimating an ecosystem model. However, if certain types of data are available, it is still possible to determine the welfare losses associated with an environmental perturbation. This can be done in a straightforward fashion provided that data on the population of the critical species, the catch of the critical species, and appropriate environmental quality variables are available. The methodology is outlined below.

The natural growth function .of equation (1) is easily estimated when written in the following form

$$F(X_t) = rX_t - (r/K)X_t^2 + \epsilon_t = \alpha_1 X_t + \alpha_2 X_t^2 + \epsilon_t \qquad (8)$$

where $F(X_t)$ is the recruitment to the stock in the year t and can be measured as $X_t - X_{t-1}$ where C_{t-} is the catch in year t-1.

An examination of the error term of this equation can lead to some important insights into the direction in which the analysis should procede. In a completely constant environment, equation (8) would be essentially an identity, with any residual due to measurement error alone or possible small random fluctuations systematically unrelated to any variable in the system. Of course when equation eight is estimated using time series data, the error term is much more encompassing than this, as the assumption of a constant environment is not valid and the error term will embody actual flucuations in K (or r) due to an environment which varies over time. Such environmental changes would include naturally generated changes due to factors such as weather, or from man-made causes such as pollution or destruction of habitat. The flucuations in K which are generated by changes in environmental parameters are exactly what must be explained in order to begin determining the benefits or costs associated with the changes in the environmental parameters. If one can obtain field observations from a series (cross-sectional or time) of unexploited subsystems of the ecosystem, then the estimation of K as a function of environmental parameters such as salinity levels, nutrient concentrations, dissolved oxygen levels, etc. is quite straight forward. In general this may prove difficult to do, because if the species of interest is really important, few if any unexploited ecosystems are likely to exist.

There is an approach which can work quite well, however, as long as the data on population, catch and the environmental variables are available. This approach would be to estimate equation (3), but rather than include the unobserved K on the right hand side, substitute some function of the environmental variables. If Q_k represents the k^{th} environmental variable, then equation (3) can be rewritten as equation (9).

$$F(X) = rX\left[1 - \frac{X}{\sum_{k=1}^{n} \delta_k Q_k}\right] \qquad (9)$$

where F(X) is approximated as $X_t - X_{t-1} + C_{t-1}$[4]. Note that the denominator of the fraction is presented as a linear function. This is done for analytical simplicity, but a nonlinear function could be easily substituted.

Equation (9) should be easily estimable using nonlinear estimation techniques. The natural growth function specified in equation (9) can then be interpreted as an equilibrium catch equation and substituted for equation (6) in the system of equations which describes the bioeconomic equilibrium for the economic activity. The inverse demand and marginal cost functions are presented as linear functions in equations (10) and (11). The three equation system ((9)-(11)) can be solved as before, and net benefits calculated in the same fashion,. The only difference is that the intermediate step of computing K has been eliminated. The original values (or naturally occuring values) of the various Q_k can be subsituted into equation (9) and the model solved to yield the net benefits associated with the economic activity. The level of a particular Q_k can then be varied and the model solved again to fired the welfare losses associated with a change in Q_k.

$$MC = a_0 + a_1 C + \sum_{i=1}^{m} \alpha_i \rho_i + a_3 X + \theta \qquad (10)$$

$$\Pi = b_0 + b_1 C + \sum_{j=1}^{h} \beta_j S_j + \phi \qquad (11)$$

Very often the ability to estimate such a model will be limited by the number of observations on population. For a variety of reasons, data on the species population is likely to be available for a much shorter period of time than for the economic variables and many of the other environmental variables. In some circumstances the time series may be so short that there are insufficient degrees of freedom to estimate equation (9). However, as long as a reasonable number of observations on population exist, it is possible to construct a fairly reliable proxy for population. This proxy is based on one already suggested in the literature, catch per unit effort. However, rather than use catch per unit effort as a proxy for X directly in the supply function (under these circumstances one would not even attempt to estimate the growth function), catch per unit effort is used to provide an estimate of population for those years for which catch per unit effort data is available, but not population data. This can be done by running a simple regression (equation 12) of population on catch per unit effort. If catch per unit effort is as good a proxy for population as has been suggested by the frequency of its use in the fisheries literature, then X should be a good approximation for X for those periods in which C and E are observed, but not X.

$$X_t = \gamma_0 + \gamma_1 \left[\frac{C_t}{E_t} \right] + \psi_t \qquad (12)$$

In many circumstances, there may no observations on population, or simply too few observations to even attempt estimating the simple regression of equation (12). Under these circumstances, the attempt to estimate the equilibrium catch equation becomes problematic [5] and a modified and less satisfactory methodology must be employed. This methodology would involve collapsing the three equaition system of equations (9)-(11) into the two equation system of equations (13) and (14). Note that the major modification has been to eliminate the unobserved X from the right hand side of the marginal cost function and substitute those environmental variables (Q) which partially determine X. With this system it is still possible to measure the change in net benefits associated with a change in a particular Q_k, however caution must be employed as the results are likely to be a reasonable approximation only for marginal changes in Q_k. The reason for this is that when the biological equilibrium is no longer explicitly considered in the solution of the overall equilibrium, an implicit assumption of biological equilibrium must be made. As changes in the environmental variables (Q) displace the equilibrium further and further from the original equilibrium, the assumption that the population remains in biological equilibrium becomes increasingly strained.

$$MC = a_0 + a_1 C + \sum_{i=1}^{m} \alpha_i \rho_i + \sum_{k=1}^{n} \alpha_k Q_k + \theta \qquad (13)$$

$$\Pi = b_0 + b_1 C + \sum_{j=1}^{h} \beta_j S_j + \phi \qquad (14)$$

Both the three equation model (equation (10)-(12)) and the modifed two equation model (equations (13) and (14)) model will be estimated for oysters in the Appalachicola Estuary in Northern Florida. The environmental variables which appear to be important in the equilibrium catch function are the flow of the Appalachicola River and the area of oyster reef. Oyster reefs are important because the oyster spat must fall on hard bottomed areas in order to survive. Oyster reefs are naturally occuring phenomena which can be augmented by the planting of discarded shells from oyster shucking operations. The proportion of total bottom area covered by reefs is largely independent of environmental variables which can be influenced by policy, but dependent on truly random events such as hurricanes which may cover existing reefs with sediment, rendering them incapable of supporting‘ oysters. River flow is important because it is linked to the salinity of the estuary, which is hypothesized to have a nonmonotonic (increasing and then decreasing) effect on oyster populations (Meeter, *et al.,* Chanley). As Atlanta (which is located upstream on the Apalachicola River) grows, a greater volume of water is removed to service the population and industry in the region. The three equation model can measure the welfare losses associated with the decline in oyster populations associated with the diversion of river flow, both at the margin and in total.However, the two equation modal can only estimate marginal changes in net social benefits.

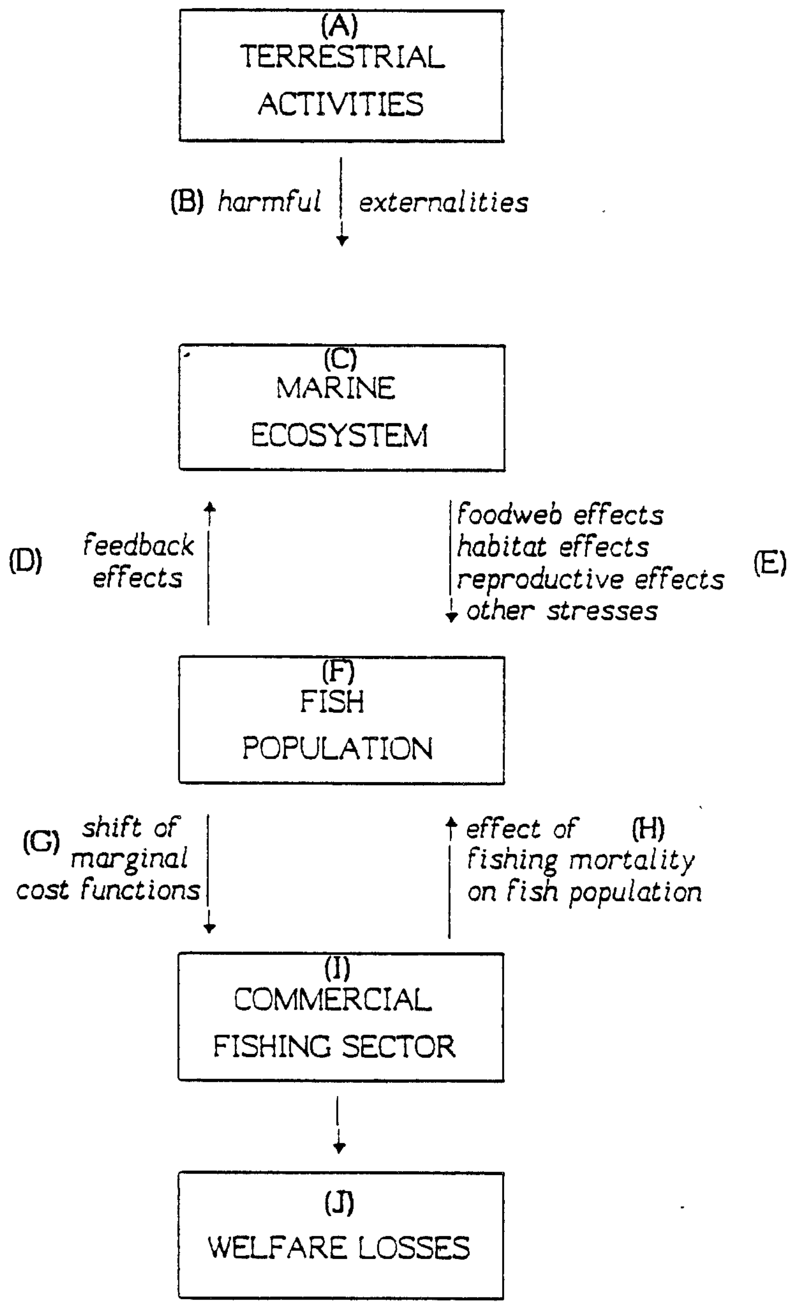
(NOTE TO DISCUSSANTS: The two equation model will be estimated in time for the workshop in June. Unfortunately, the data on osyter populations will not be available until late summer or early fall, so I will not be able to present those results, us I had hoped.)

CONCLUSIONS

It is possible to estimate the welfare losses-associated with terrestrial pollution of marine ecosystems if the effect of the environmental change on the parameters of the equilibrium growth function can be specified. The ideal way to do this is by integrating the economic model with a detailed ecosystem model, but these detailed ecosystem models are often unavailable. In the absence of such a mdoel it is possible to specify the welfare losses provided that data on the environmental variables and the populations of critical species are available. This is done by including these environmental variables as explanatory variables in the equilibrium catch equation. When it is not possible to estimate an equilibrium catch equation (no population data) it is still possible to approximate these welfare losses by including the environmental variables as explanatory variables directly in the marginal cost functions. This model, however, is less satisfactory because it does not explicitly model the biological equilibrium of the species of interest.

An important area of future research is to test these (and competing) methodologies in as many different applications as possible. However, the ability to engage in this type of analysis is constrained by data limitations. In order to develop better policy, it is neccessary to develope more and better data. Of particular importance is time series data on the levels of species' populations and environmental quality variables which are of policy relevance.

Figure 1.


(A)
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(B) harmful
externalities
(C)
MARINE
ECOSYSTEM
(D)
feedback
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habitat effects
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(H)
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(I)
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FISHING SECTOR
(J)
WELFARE LOSSES

Fig 2

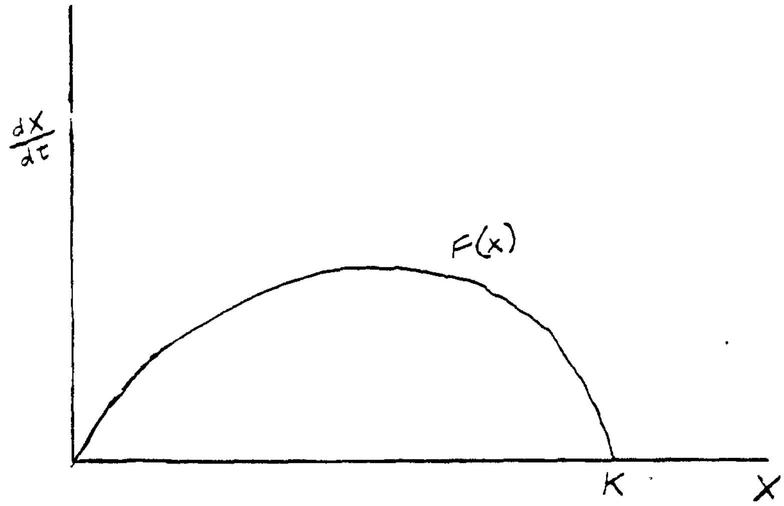


Fig 3

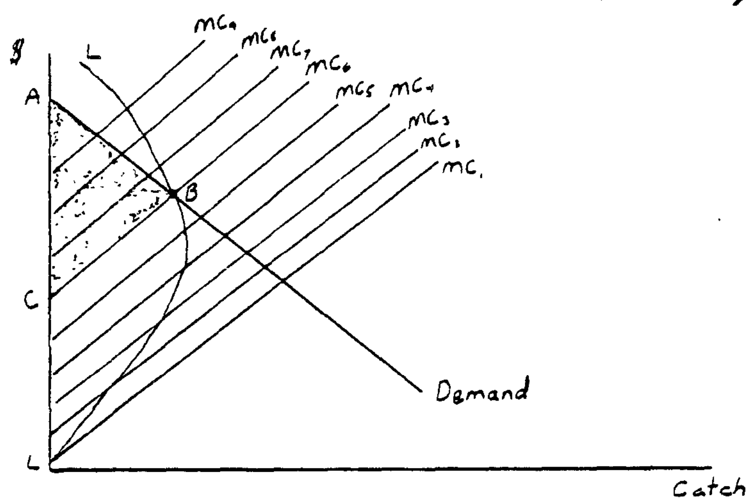


Fig 4

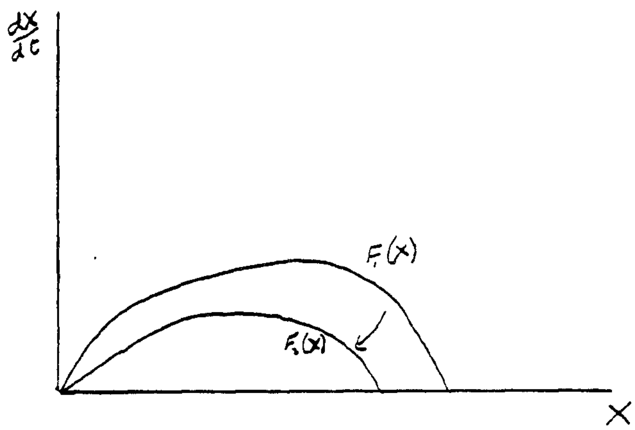
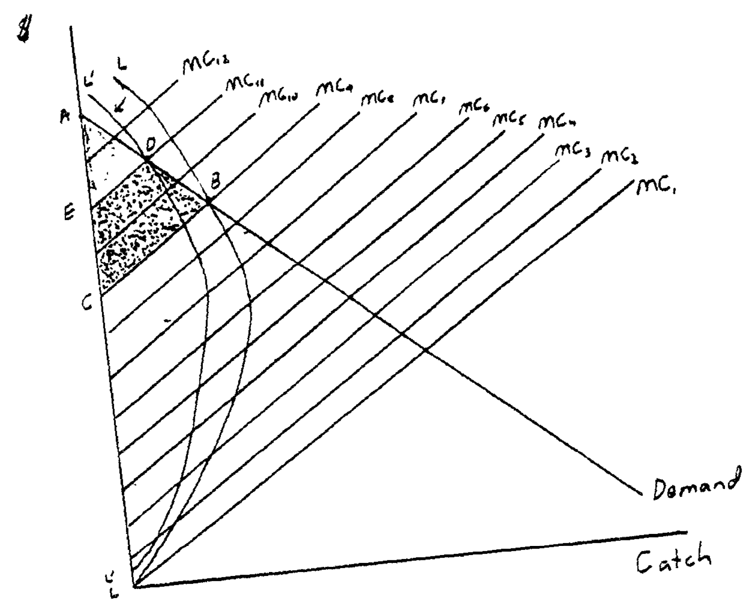


Fig 5



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